BLM

Southern Nevada Complex Emergency Stabilization and Rehabilitation Final Report

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Executive Summary

BACKGROUND

Between June 22, 2005 and July 10, 2005, the 11 fires that make up the Southern Nevada Complex (SNC) burned 739,037 acres in southeastern Nevada, southwestern Utah and northwestern Arizona. The Ely District Bureau of Land Management (BLM) manages 597,096 acres across four of the SNC fires—the Delamar fire (168,007 acres), the Duzak fire (214,038 acres), the Halfway fire (66,487 acres), and the Meadow Valley fire (148,564 acres). The location and spatial extent of the Ely BLM burned areas are displayed in Figure 1. The Southern Nevada Complex Emergency Stabilization and Burned Area Rehabilitation (SNCESBAR) Final Report looks at the Emergency Stabilization and Rehabilitation (ESR) efforts on the Ely BLM managed lands.

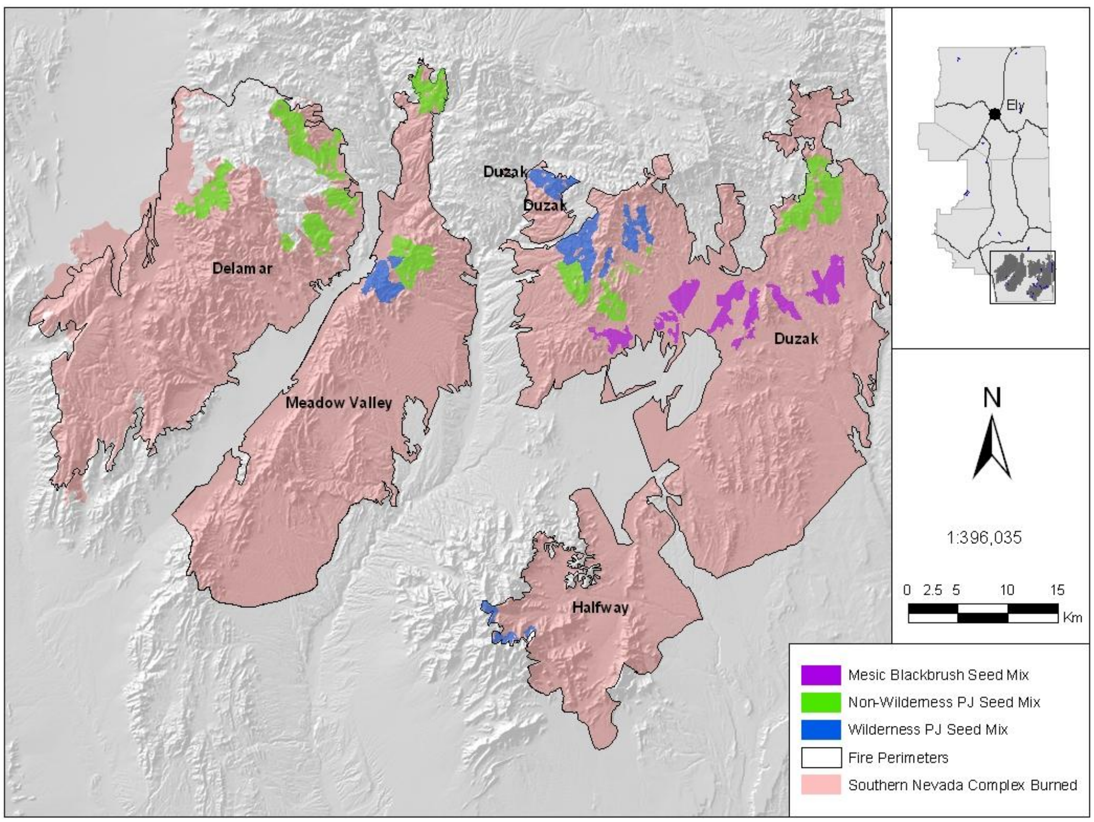

Figure S-1. Location and spatial extent of the Southern Nevada Complex fires.

The SNC fires were initiated on June 22, 2005 by dry lightning storms and spread quickly due to high fuel loads and winds. Heavy rains during the previous winter and spring (resulting in more than 400% of normal rainfall from January through April 2005) allowed for unusually high production of non-native annual grasses (predominantly red brome and cheatgrass) that were able to carry fire through areas of normally sparse vegetation. These

fine fuels served as ladder fuels between grasses, forbs, shrubs and trees of at least 12 vegetation communities in the Mojave Desert and Great Basin regions.

The SNC fires exhibited extreme fire behavior. Initial fire starts grew rapidly. Many times starts had grown to 1000 acres or more before initial attack. The situation was complicated by restrictions to suppression activities as a result of environmental concerns about endangered desert tortoise habitat. Initially this limited the use of engines and dozers to existing roads. The most common suppression tactics were retardant and water drops, backfires, and using roads and geographic features. Hand line construction was limited by fast moving fire and high rates of spread.

The eleven SNC fires burned nearly 740,000 acres in the Mojave Desert during the summer of 2005, more than the nearly 722,000 acres of fire this desert experienced in the preceding 25 years combined (Brooks and Matchett 2006). It was a significant ecological event for the Mojave Desert and presents many land management challenges in southern Nevada and for the Ely District BLM.

GEOGRAPHIC SETTING

The SNC fires burned primarily in the Basin and Range Physiographic Province, in relatively narrow north-south-trending mountain ranges separated by wider sediment-filled basins. The exception to this trend is the Clover and Mormon mountain ranges. The Clover Mountains trend east west, forming a boundary between the Great Basin desert to the north and the Mojave Desert to the south. The Mormon mountains are almost circular. SNC elevations range from about 2,000 feet in the southeastern valleys to over 8,500 feet in the North Pahranagat Range.

Average annual rainfall is 4-20 inches across the SNC depending on elevation, and precipitation is highly variable from one year to the next. The SNC experiences an arid to semi-arid climate. Summers tend to be hot, dry and windy, and freezing temperatures often occur during the winter, particularly in higher elevation regions. Lightning events are common, particularly in July and August during the monsoon season, and are a primary source of wildfire ignition.

Surface water resources in the area are limited and groundwater is scarce. Most stream channels within the SNC are ephemeral, flowing only during and immediately after rainfall events. Major flooding is caused by winter storms or rain-on-snow events in the higher elevations. Flash flooding can occur in any area at any time of year but is most probable during the summer and fall. Meadow Valley Wash is the only perennial stream within the SNC boundaries.

The dominant soil orders found within the burned areas are mineral soils low in organic matter with layers that are highly variable in thickness, texture, rock fragment content, and physical and chemical properties. Processes involving sand and dust transport play an important roll in shaping the landscape and the ecosystem of the Mojave.

A variety of vegetation communities occur within the boundaries of the SNC (Fig. 2). Their distribution is greatly influenced by rainfall and topography. The southern end of the fires occurred within the Mojave Desert of the Sonoran Basin and Range MRLA, while the northern end of these fires occurred within the Southern Nevada Basin and Range MRLA (NRCS 2002 referenced in: USDI National Interagency BAER Team 2005).

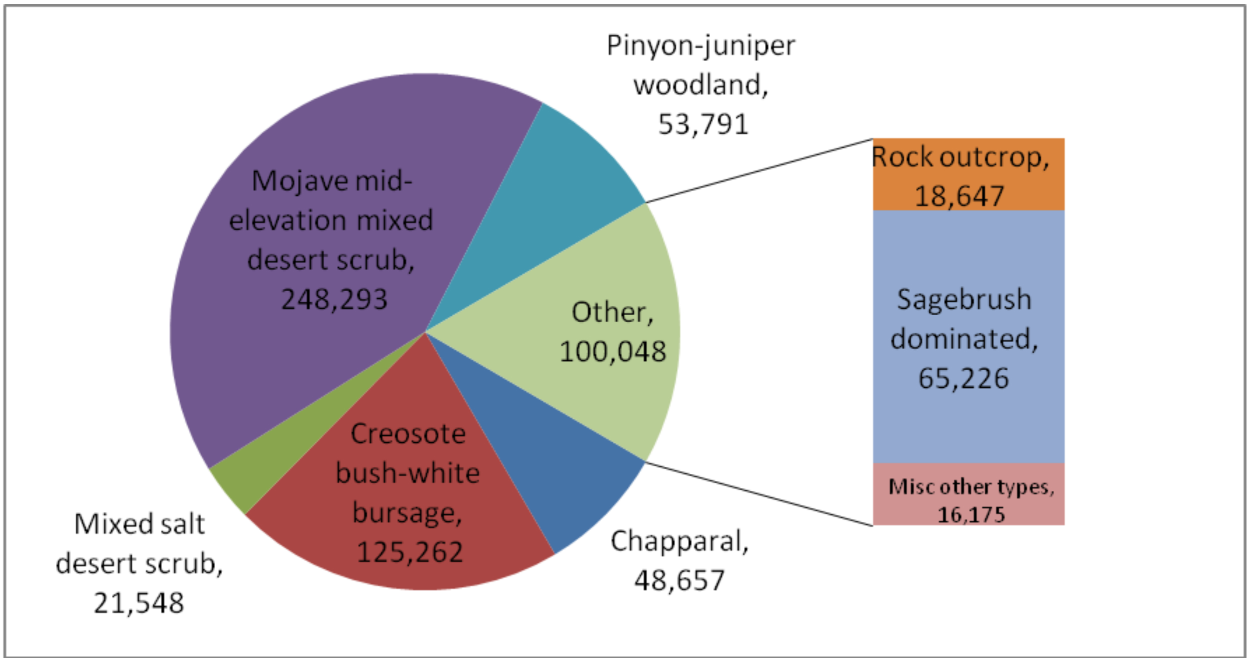

Figure S-2. Vegetation types burned by the SNC fires in the Ely District. Numbers reflect acres burned.

Additional physiographic information on the SNC fires can be found in Chapter 1 of the full report.

BURNED AREA ASSESSMENT

Prior to the SNC, the largest fire the Ely BLM District had experienced was the 1999 Delamar Fire that burned 22,592 acres. Due to the size and scope of the SNC fire, and BLM staff limitations both locally and regionally, a national Burned Area Emergency Response (BAER) team was called in to assist with the initial assessment of SNC fire damage to BLM lands and development of the Emergency Stabilization (ES) plan.

BAER Team resource specialists were tasked with assessing potential risks to life, property, and natural and cultural resources from the SNC fires and identifying potential treatments. Resources assessed included: vegetation, soils and watershed, wildlife, cultural, operations, and recreation.

Resource Assessment Findings

BAER Team vegetation specialists determined that a majority of the SNC burned area experienced low to moderate vegetation mortality (71%) with some areas of unburned to very low (15%), moderate (13%), or high mortality (1%). Vegetation mortality rates within these classes ranged from 6 to 60%, 0 to 5%, 61 to 85% and 86 to 100% respectively. While fire intensity varied throughout the burned area, the rapid rate of fire spread resulted in consumption of most of the grasses and herbaceous species, and some of the shrubs and scattered trees.

The majority of the high vegetation mortality class fell in those plant communities of the Mojave Desert that are not adapted to fire. Of particular concern was approximately 300,000 acres of blackbrush (*Coleogyne ramosissima*)-dominated communities burned in the SNC fires. While many species can resprout following fire, blackbrush is not a fire-tolerant species and generally never resprouts. In addition, much of the burned area occurred in rangelands, where non-native invasive species were present pre-fire, and in pinyon-juniper woodlands which often support very little herbaceous understory. Across the SNC, the spread of noxious and non-native invasive species was anticipated.

BAER Team soil and watershed specialists found that most of the burned area experienced low soil burn severity with some areas of moderate to high severity. [Soil burn severity relates specifically to effects of the fire on soil conditions (e.g., amount of surface litter and duff, infiltration rate, erodibility, and soil structure).] Roads on federal lands in the SNC were considered to be at a slight increased risk of flooding or being inundated with flood debris during intense storm events post-fire, and there were increased concerns for high dust concentrations that could create near “white-out” conditions. However, no significant values at risk from post-fire erosion or runoff were identified within the burned areas of the four SNC fires managed by the Ely BLM.

Many potential risks to life, property, and natural resources were identified downstream of the Duzak and Meadow Valley fires in large part because these fires burned watersheds that drain into Meadow Valley Wash, Beaver Dam Wash and the Virgin River, the majority of whose channels were rearranged by floods resulting from a January 2005 rain-on-snow event, increasing concerns for cumulative effects from post-fire flooding.

BAER Team wildlife specialists anticipated long term effects on livestock grazing from the SNC fires. Areas of particular concern were the Mojave Desert ecosystem where low precipitation results in lengthy recovery times and pinyon-juniper woodlands which often support very little herbaceous understory. Both of these ecosystems were considered susceptible to a post-fire conversion to non-native annuals that offer little forage. The SNC fires burned across 27 allotments and four Herd Management Areas (HMAs) in the Ely District.

There are seven federally listed species (T&E) and one Candidate species that occur within the SNC or downstream receiving water bodies and riparian areas. Only the desert tortoise was expected to be adversely impacted by the fires. In addition, the SNC further diminished important mule deer and elk habitats, especially in blackbrush and pinyon-juniper communities in mid-to upper elevations. These habitats are declining in amount and quality range wide, due to drought, fire and fragmentation.

There are 148 cultural resource sites located within the SNC lands managed by the Ely BLM. BAER Team cultural specialists found no indication that any of the sites were impacted by the fire or suppression activities and none appeared to be threatened by erosion. However, there were concerns for site looting and off-highway vehicle (OHV) damage.

BAER team operations specialists identified an increased chance of public exposure to potential hazards from the SNC fires (e.g. abandoned mines, hazardous materials, road washouts, and public safety sign damage) and an increased workload associated with implementing the SNC ES Plan.

No developed recreation sites exist within the SNC burned areas managed by the Ely BLM District. However BAER Team recreation specialists found that many opportunities for dispersed, primitive, and unconfined forms of recreation were impacted by the fires, including quail hunting, pine nut collecting, heritage tourism, geocaching, hiking, OHV touring, and to a lesser extent some desert bighorn sheep, mule deer, and small mammal hunting.

A more detailed representation of resource assessment findings can be found in Chapter 1 of the full report.

TREATMENT RECOMMENDATIONS AND IMPLEMENTATION

In response to resource assessment findings, BAER Team resource specialists identified 27 individual specifications/treatments for the 597,096 burned acres within the Ely BLM District. Due to concerns about long-term blackbrush loss, resource specialists at the Ely BLM wrote a Burned Area Rehabilitation (BAR) Plan in the fall of 2005 focused on reestablishing blackbrush.

A majority of the proposed and implemented ES and BAR specifications/treatments are considered minor for the purposes of reporting and analysis. These include: Native American Consultation, Implementation Leaders, Wilderness Access Hand Seeding, Post-Emergent Herbicide Application Along Roads in Burned Area, Lop and Scatter, Model Flow and Sediment Delivery to Meadow Valley Wash and Beaver Dam Wash, Stateline Boundary Exclusion Fencing, Exclosure Fencing, Known Cultural Site Assessment, Wild Horse and Burro Gather, Public Safety Hazard Assessment, Replace and/or Install Public Safety Signs, Temporary Administrative Vehicle Route Closure, Wilderness Track Seeding, Hand Seeding in Desert Tortoise ACEC Habitat, Intensive Rehabilitation Islands, Wildlife Water Source Rehab, Noxious Weed and Invasive Plant Control and Revegetation, Seed Collection for Desert Tortoise Areas of Critical Environmental Concern, Minor Facilities Repair and Replacement, Wild Horse Census, and Wildlife Water Developments Repair. Not all of the treatments recommended by the BAER team were approved or funded. See Chapter 2 in the full report for detailed information on approval and implementation of the treatments above.

The majority of the SNCESBAR Final Report (Chapters 3-9) focuses on three treatments: 1) the post-fire hand seeding treatments in desert tortoise critical habitat; 2) aerial seeding treatments in blackbrush and pinyon-juniper communities; and, 3) the analyses of treatment effectiveness from monitoring data collected between 2006 and 2008. These chapters are summarized below.

Effectiveness of Post-Fire Seeding in Desert Tortoise Critical Habitat Following the 2005 Southern Nevada Fire Complex

The Southern Nevada Fire Complex burned more than 32,000 acres of designated desert tortoise Critical Habitat and an additional 403,000 acres of Mojave Desert habitat characterized as potentially suitable for the tortoise (Fig. 3). To accelerate the re-establishment of plants commonly used by tortoises for food and shelter, the BLM seeded native annual and perennial (grass, forb and shrub) species in burned desert tortoise Critical Habitat in 2005 and 2006.

The United States Geological Survey (USGS) was tasked with monitoring vegetation and tortoise responses for three years after burned tortoise habitat was seeded to determine whether: 1) non-native annual plant production was reduced, and native annuals increased on

sites seeded with native species; 2) perennial plant density and canopy cover were augmented by seeding; and, 3) tortoise activity, as indicated by detection of recent tortoise sign (live tortoises, active burrows, fresh scat, and tracks), increased in seeded areas. They also monitored seed banks to determine if viable seeds from the seed mix persisted one and two years following application and, they monitored monthly precipitation to evaluate vegetation responses among the broadly distributed sites on the SNC.

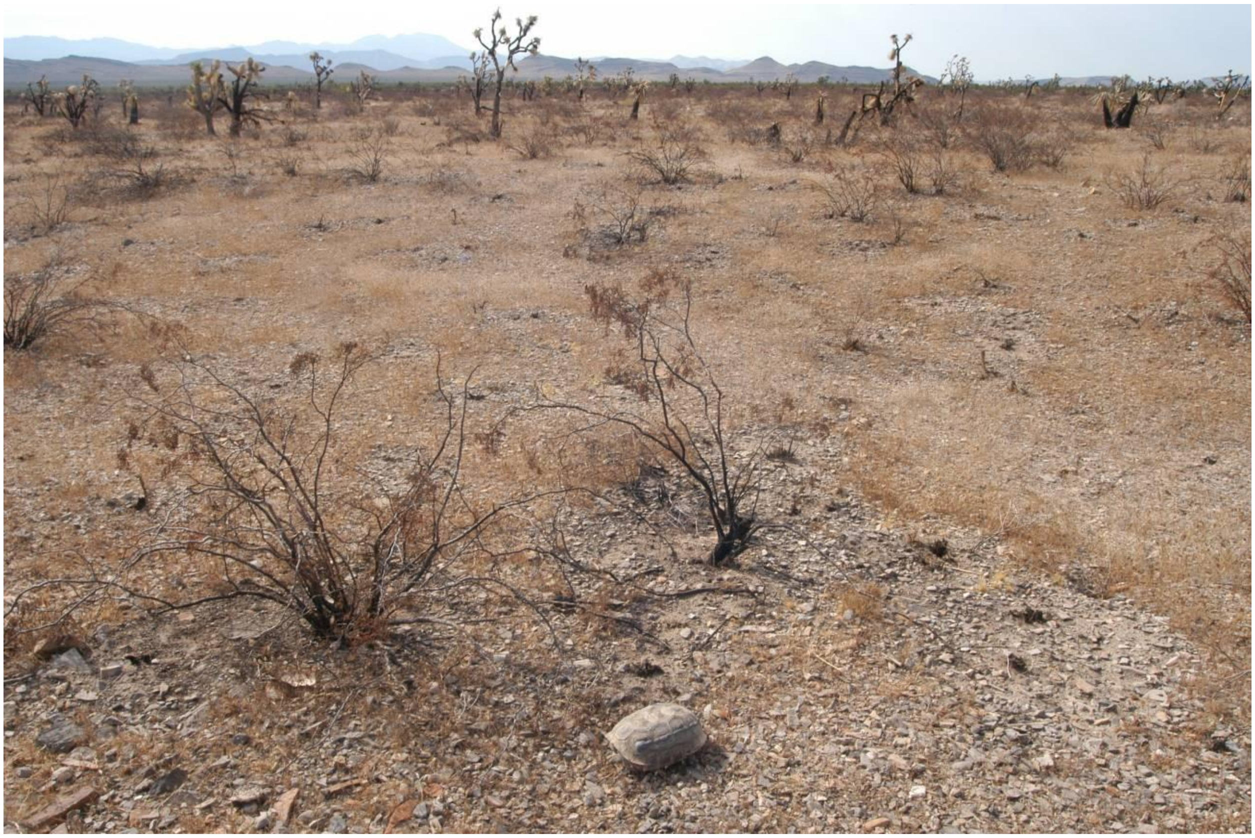

Figure S-3. A desert tortoise is faced with a dramatically altered habitat following the 2005 Dry Rock Fire (Photo: L. A. DeFalco).

Within the three-year ESR monitoring period the USGS found that seedling densities of seeded perennial species were 33% higher in seeded areas than in unseeded areas, particularly for the disturbance-adapted desert globemallow (*Sphaeralcea ambigua*) and desert marigold (*Baileya multiradiata*) which displayed rapid seedling emergence and establishment. They also found that seeding augmented perennial seed banks by four- to six-fold within a year of seed applications compared with unseeded areas, demonstrating that seeding increased the long-term recovery potential of seeded burn sites. Seeded annuals, in contrast, did not increase significantly in seed banks or biomass production, likely due to low seeding rates of these species. Production of non-native annuals that helped carry the fires was not reduced by seeding efforts but instead was strongly correlated with site-specific rainfall, as were native annual species.

Tortoises were observed moving along fire boundaries, using vegetation and burrows as shelter in unburned habitat while foraging and basking within burned areas where more herbaceous forage is available. Tortoise activity was not enhanced in seeded areas during the three years following seeding.

The USGS suggested that maximizing seeding rates and reducing seed residence time with seasonally-appropriate application may improve establishment of native annuals and fast-establishing perennials and deserves further research. They also noted that desert tortoise activity in burned areas was likely hindered during the three year monitoring period by lack of shelter sites, and that longer-term monitoring of both canopy cover and tortoise activity is necessary to determine the effectiveness of seeding for tortoise habitat restoration.

See Chapter 3 in the full report for more details on the effectiveness of post-fire seeding in desert tortoise critical habitat following the SNC fires.

Delineation of Final Aerial Seeding Polygons and Sampling Design for Aerial Seeding and Natural Regeneration Treatment Effectiveness Monitoring

The BAER team called for aerially seeding a total of 47,000 acres on the four Ely District SNC Fires, in order to stabilize soils and control the spread of invasive non-native species. They delineated potential seeding polygons covering more than 82,000 acres (32,200 ha) in pinyon-juniper and close to 50,000 acres (20,000 ha) in mesic blackbrush (Fig. 4). They also recommended monitoring ES treatment effectiveness for vegetation recovery, seeding success, and noxious and invasive non-native weed expansion.

Using the BAER team potential seeding areas, digital aerial photographs, ground reconnaissance and some GIS data layers provided by the United States Geological Survey Earth Resources Observation and Science Center (USGS EROS), BLM and Eastern Nevada Landscape Coalition (ENLC) staff delineated 30 seeding polygons, covering 47,000 acres in pinyon-juniper and mesic blackbrush communities, across all four fires. Thirteen polygons were seeded with a Non-Wilderness PJ Seed Mix, eight with a Wilderness PJ Seed Mix, and nine with a Mesic Blackbrush Seed Mix. Seeding polygons ranged in size from 103 acres to 5,589 acres.

The USGS Western Ecological Research Center (WERC), BLM, and ENLC designed three primary sampling methodologies to monitor treatment effectiveness, following the application of the aerial seeding treatments. These methodologies were brushbelt (BB) macroplots (designed to specifically compare seeded and unseeded conditions in demonstration plots), additional aerial seeding coverage (AA) macroplots (designed to maximize sampling coverage of environmental heterogeneity of the aerial seeding polygons including topographical variables, soils, and pre-fire vegetation) and, natural regeneration plots (used to evaluate the majority of the burned area that was not seeded). In addition to the

three primary sampling methodologies, monitoring crews also conducted qualitative assessment write-ups and ocular cover estimates to increase coverage of non-seeded burned areas and help ground truth remote sensing tools being developed and evaluated for their utility on the SNC and on fires in rangeland/arid ecosystems more generally.

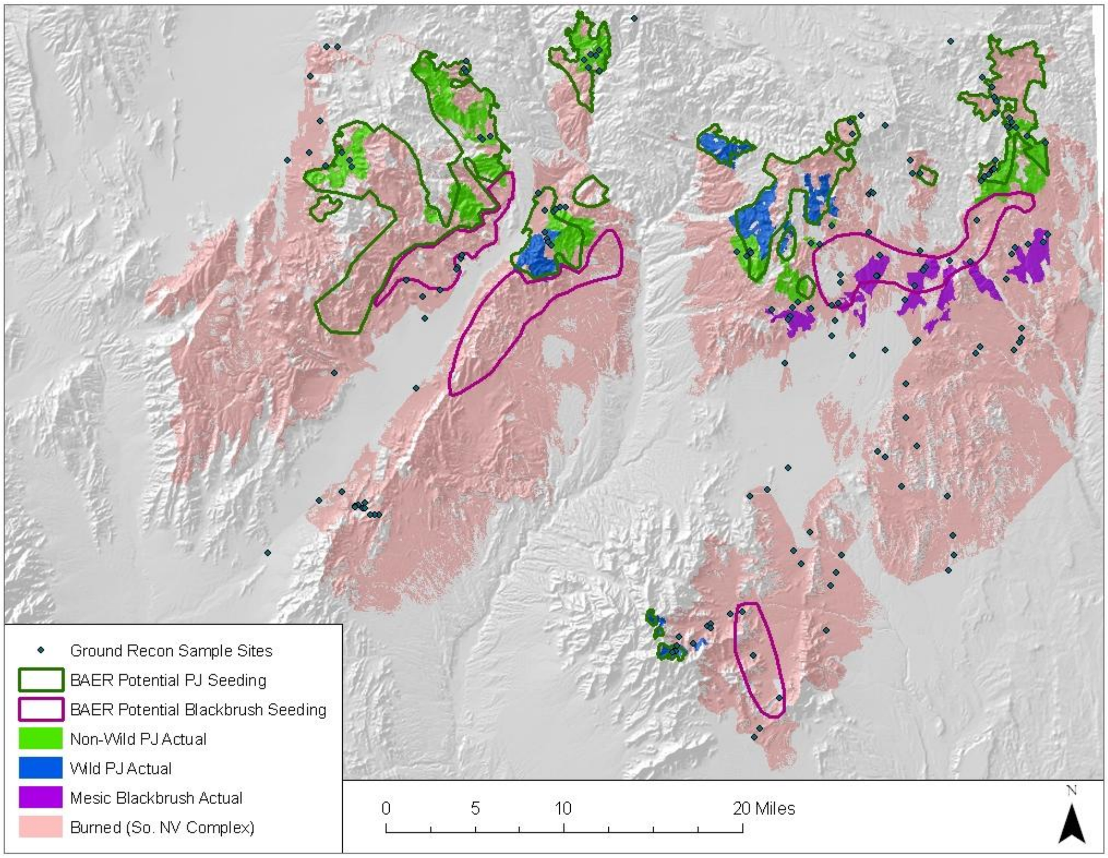

Figure S-4. Comparison of BAER team potential seeding areas to polygons actually seeded. Reconnaissance photo points were used to delineate actual seeding polygons from the BAER team potential seeding areas.

See Chapter 4 in the full report for more details on the delineation of the final aerial seeding polygons on the SNC and sampling design for aerial seeding and natural regeneration treatment effectiveness**.**

Establishment of Aerial Seeding Treatments in Blackbrush and Pinyon-Juniper Sites Following the 2005 Southern Nevada Complex

The USGS was also tasked with evaluating the level of seeded species establishment in the 47,000 acres of mesic blackbrush and pinyon-juniper aerially seeded within the four Ely BLM managed SNC fires. Objectives were to determine if: (1) the seeded species met planned seeding density objectives; (2) seeded species diversity, frequency, and density increased; and, (3) seeded species diversity, frequency, and density differed among

vegetation types and between seeded and unseeded areas over the three year monitoring period. The USGS also discussed establishment patterns and their implication for future post-fire seeding projects in similar ecotypes of the Mojave Desert.

Over the three year ESR monitoring period, the USGS found that the seeded species comprised a minor component of the post-fire vegetation communities. On average, densities of seeded species were 2-3 orders of magnitude lower than the desired levels of three seeded perennial species m^{-2} in mesic blackbrush and five seeded perennial species m^{-2} in pinyon-juniper. Frequency, density and diversity of seeded species were greater in the two pinyon-juniper communities, especially non-wilderness pinyon-juniper, compared to mesic blackbrush. This was not restricted to seeded plots though; rather, it was an overall effect of vegetation type. Although density of the seeded species was low, there was some evidence that aerial seeding increased establishment of the seeded species. There was, however, no consistent trend over time in establishment and persistence of seeded species within the plots.

Sixteen species were used in the pinyon-juniper and mesic blackbrush seed mixes. Among the seeded species that were detected, *Elymus elymoides*, *Poa secunda*, and *Agropyron cristatum* were the most widespread. Several species also appeared to have a much higher chance of establishment in at least one of the vegetation types evaluated. Notable examples for mesic blackbrush included *Achnatherum hymenoides*, *Sanguisorba minor*, and *Sporobolus cryptandrus*. For non-wilderness pinyon-juniper, *Agropyron cristatum* appeared in more seeded than unseeded, and for wilderness pinyon-juniper *Achnatherum hymenoides* and *Pleuraphis jamesii* appeared in more seeded than unseeded plots. More information is needed on how establishment rates vary with the seasonal timing of seeding, species composition of seed mixes, application rates, and among vegetation types and years of contrasting climatic conditions. However, these species should be considered for systematic evaluations of their likelihood of establishment in the Ely BLM District.

Only the BB plots data (designed to specifically compare seeded and unseeded conditions in demonstration plots) was used in the USGS evaluation of seeded species establishment. Analyses were further confined to species richness, frequency of occurrence, and density of the seeded species. This was ostensibly so the analyses could be focused on comparisons of seeded species establishment in seeded versus unseeded areas, and so the USGS could evaluate how these comparisons varied among vegetation types. It should be noted that Ely District ESR staff do not agree with the USGS approach to analyzing the aerial seeding monitoring data particularly the exclusion of much of the collected data.

See Chapter 5 in the full report for more details on the establishment of aerial seeding treatments in blackbrush and pinyon-juniper sites following the SNC.

Vegetation Trends following the 2005 Southern Nevada Fire Complex

The USGS was also tasked with analyzing the spatial and temporal plant succession patterns in the four Ely District SNC Fires, especially in relation to dominance by non-native grasses and forbs. Their primary focus was on distribution and abundance patterns of cheatgrass (*Bromus tectorum*), red brome (*Bromus rubens*), and red-stemmed filaree (*Erodium cicutarium*). Specific objectives were to analyze: (1) the relationship of vegetation structure and species composition with topographic, disturbance, and rainfall variables; (2) the spatial and temporal patterns of abundance of cheatgrass, red brome, and *Erodium* in different vegetation communities, as well as their relationships to gradients in elevation, precipitation, and disturbance; and, (3) the correlation between patterns of native herbaceous and woody species with abundance of cheatgrass, red brome, and *Erodium*, and discuss how this is likely determining succession trajectories in these post-fire communities.

Analyses were focused on four vegetation types arranged along an elevation gradient in the Mojave Desert. They included wilderness and non-wilderness pinyon-juniper vegetation types which occur at the highest elevations, followed by the mesic blackbrush vegetation types at middle elevations, and the thermic blackbrush and upper elevation creosotebush scrub (referred to as the "natural regeneration" vegetation type) at the lower elevations. The USGS found that precipitation had an extremely important influence on succession patterns and species composition in the vegetation communities studied. It was the primary factor responsible for higher levels of recruitment of woody species and native bunchgrasses and increased species richness and stem density of native and non-native annual species. Precipitation was not, however, an important factor in regeneration of woody stems. It was not just the amount of rainfall that occurred in a given year that influenced species composition, but the timing of it as well. Species composition in all of the vegetation types was dramatically affected by the amount of rainfall that occurred in the early or latter periods of the wet season.

Density of shrubs and trees (resprouts and mature individuals) was higher in pinyon-juniper communities than natural regeneration and mesic blackbrush communities in all years. Burn severity was not consistently associated with vegetation responses. The densities of some species were highest at low fire severity values, whereas others were highest at high fire severity values.

Although they comprised less than 10% of the herbaceous flora, stems of non-native grasses and forbs dominated post-fire vegetation communities on the SNC. Collectively cheatgrass, red brome, and *Erodium* made up 90% or more of the stems in the BB monitoring plots, and one or more of these species dominated the post-fire flora regardless of year, elevation zone, vegetation type, or burned area. These species, however, showed very different spatial and temporal patterns. Red brome and *Erodium* dominated lower elevation

communities (with peak densities between 800 and 1200 meters) and had higher abundances in areas with low or, at most, average amounts of precipitation. In contrast, cheatgrass dominated higher elevation communities (with a peak density at 1800 m), and its abundance increased dramatically as precipitation increased.

There was evidence that competition from these three, non-native species was intense enough that it was likely suppressing woody regeneration (by both seedlings and resprouts) and species richness of native perennial grasses. Only on the rare occasion that density of cheatgrass, red brome, and *Erodium* was low and rainfall was high were seedlings of woody perennials abundant. However, competition from the Brome grasses and *Erodium* was not a factor in suppressing regeneration of native herbaceous species. Species richness of native annual forbs and density of native perennial forbs actually had a positive relationship with density of cheatgrass, red brome, and *Erodium*. [This likely reflected a common response to an increase in resource availability, regardless of whether species were native or not (Stohlgren et al. 1999).]

The USGS concluded that suppression of woody regeneration could be enough to allow the SNC communities to remain dominated by non-native annual species over time. Alternatively, if these areas do not undergo further disturbances for several decades or regeneration or woody species is not impeded by grazing, succession could lead to shrub-dominated communities. The USGS found it difficult to say whether richness and stem densities of other species would increase if stem density of the two Bromes and *Erodium* was substantially reduced.

See Chapter 6 in the full report for more details on vegetation trends following the SNC.

General Vegetation Trends and Seeded Species Establishment: A Descriptive Analysis Using Data from AA Macroplots

Monitoring of ESR treatments is usually conducted for three growing seasons post-fire. In situations where monitoring funds are limited, and in large fire years where burned acreage is extensive, trade-offs must be made between more intensive monitoring strategies that provide a detailed examination of why ESR treatments succeed or fail and more extensive monitoring strategies that provide coarse information on what conditions are found on the ground over a large area. The BLM utilized both more intensive and more extensive monitoring strategies on the SNC fires. The cover data collected from the AA plots is an example of the latter and was used to capture the environmental heterogeneity present across the SNC over the first three growing seasons.

The AA plot data revealed a landscape dominated by a mix of different plant types, including both perennials and annuals and natives and non-natives. The data also drew a

fairly strong distinction in plant dominance between burned higher elevation pinyon-juniper woodlands and burned mesic blackbrush communities. At the higher elevations, a mix of different plant guilds dominated, and most sites included a co-dominant perennial. In many cases, this was due to resprouting shrub species (e.g. *Quercus turbinella*, *Garrya flavescens*, *Amelanchier utahensis*, etc.). At lower elevations sites, in mesic blackbrush, the landscape was dominated primarily by the non-native forb *Erodium*. These lower elevations also tended to have a perennial component, including resprouting shrubs (e.g. *Yucca baccata* and *Purshia* glandulosa), and in some areas perennial grasses such as *Aristida purpurea* were returning in high abundance by year 3 post-fire.

In general, seeded species are only establishing at very low densities on the SNC. The exception is a few localized areas in which crested wheatgrass (*Agropyron cristatum*) is found at high densities in the Clover Mountains portion of the Non-Wilderness PJ Seed Mix. Although, seeded species had increased in presence over the three-year period across all three seed mixes (Fig. 5), they had not provided meaningful competition against non-native annual grasses or forbs as of year 3 post-fire.

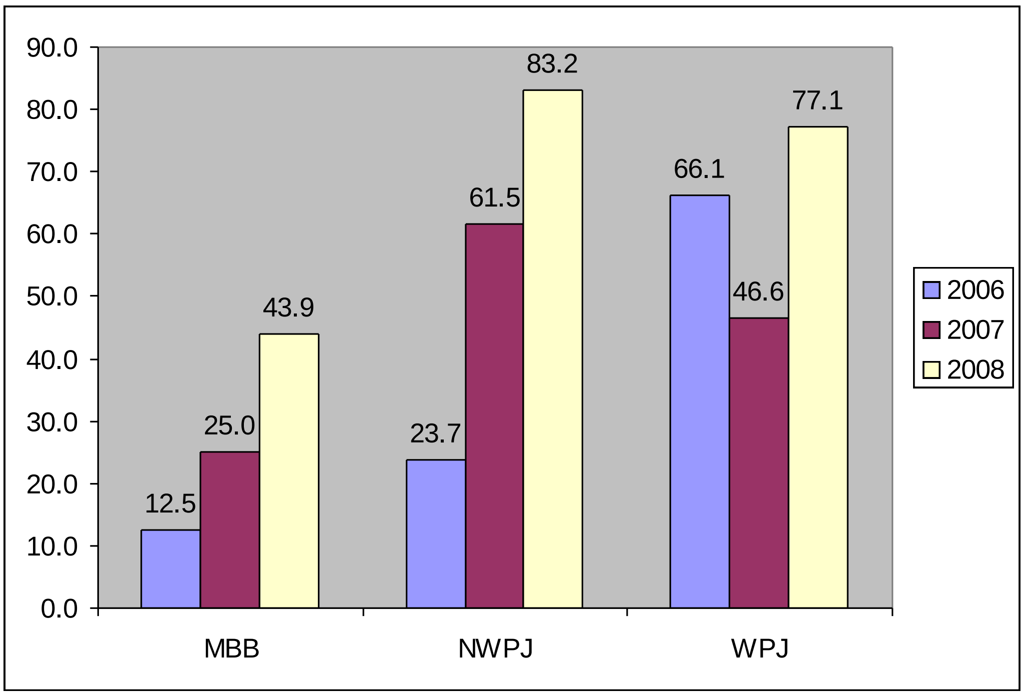

Figure S-5. Percentage of sampled AA macroplots with seeded species present within the Mesic Blackbrush Seed Mix (MBB), Non-Wilderness PJ Seed Mix (NWPJ), and Wilderness PJ Seed Mix (WPJ), 2006-2008.

Over the three-year monitoring period, annual grass dominance declined in both pinyon-juniper and mesic blackbrush sites. In the PJ seed mix polygons, there was a shift from areas dominated solely by annual grasses to areas dominated by a mix of perennials and

exotic annuals. In the lower elevations, this shift was more from annual grasses to *Erodium* dominance. It is very difficult to predict what the future post-fire plant community makeup will look like on the SNC. This is due to the high variability of dominance within the burned areas and the drastic changes from year to year. Many of the perennial species establishing in both of the PJ seed mixes are resprouters and are fairly resilient to fire. It is possible that these species are increasing in abundance, in the absence of the non-resilient species which cannot return as quickly.

See Chapter 7 in the full report for more details on general vegetation trends and seeded species establishment using data from the AA macroplots.

Soil Erosion Risks Following the 2005 Southern Nevada Fire Complex

One of the primary land management concerns in the Mojave Desert is the potential for increased dominance by non-native annual grasses following wildfires. These concerns are largely focused on competition of non-native grasses with native plants, their effects on fire regimes, and their cumulative effects on wildlife habitat (Brooks and Pyke 2001, Brooks and Esque 2002). Their relationship to soil erosion remains largely unevaluated in the Mojave Desert. The USGS evaluated the indirect evidence for fire effects on soil erosion potential on the SNC, using cover data from the BB macro-plots (including basal gap, living perennial and annual vegetation canopy cover, and litter) collected in postfire years 1, 2, and 3 in pinyon-juniper and mesic blackbrush communities.

Analysis of the data revealed no notable fire effects on basal gaps between perennial plants, which already comprise about 98% of the total ground surface in unburned areas. Any post-fire basal gap changes probably had a negligible effect on soil erosion potential. Annual plant cover was significantly reduced in pinyon-juniper communities, but not in mesic blackbrush, in year 1 post-fire. Based on current and past results, the USGS suggests that post-fire reductions in total annual plant cover may increase soil erosion potential from flowing water during the first year post-fire (at least in the higher elevation pinyon-juniper zones), but probably not beyond the first year. Declines in perennial cover were more notable, especially in blackbrush vegetation, as were declines in litter and duff cover particularly in pinyon-juniper vegetation. Reduced perennial cover has major implications for wind erosion (Herrik et al. 2005b). Although quick recovery of annual plant cover may help mitigate some of the effects of perennial cover loss, annuals cannot replace the coarse physical structure and windbreaks that only perennial plants [(which may take many decades to re-establish following Mojave Desert fires (Brooks and Minnich 2006)] provide. The loss of litter in pinyon-juniper communities can affect both wind and water erosion potential post-fire. The only litter recovery observed by year 3 post-fire was in blackbrush where litter from non-native annual grasses showed increases.

See Chapter 8 in the full report for more details on soil erosion risks following the SNC.

Remote Sensing Assessments and Application

Remote sensing technologies were first utilized for the SNC fires by the BAER team in July 2005 to generate the final soil burn severity map and several subsequent burn area analyses. The Ely BLM requested a more in-depth assessment of remote sensing capabilities for a range of post-fire uses, including assessing pre- and post-fire vegetation changes, selecting seeding sites, and assessing post-treatment vegetation changes.

The SNC geospatial database subsequently created allows the evaluation of overall post-fire vegetation greenness and estimation of recovery in terms of the “return” to pre-fire or “background” greenness levels. Ely BLM and ENLC staff used the greenness and burn severity map products to assist in making seeding treatment location decisions for the SNC and, along with numerous other GIS data layers, overall treatment decision making.

More evaluation is required, but Landsat 30 meter data appear to be sensitive to seasonal fluctuations of annual grasses and perhaps perennial plants in the vicinity of the SNC. These data also appear to have potential for monitoring greenness change within seeded and non-seeded paired-plots. However, further study is needed in treatments exhibiting significant seeded species establishment, in order to determine how sensitive Landsat 30 meter data is to vegetation greenness increases and seeding effectiveness in rangeland/arid ecosystems. Landsat 30 meter data and related map products may also prove valuable to land managers making “grazing or range readiness” determinations after large fires with limited quantitative information at a landscape scale. However, it should be noted that this estimation technique does not specifically take into account how vegetation may have changed in composition or structure over time. Rather, it is just a comparison of the overall vegetation greenness to previous (pre-fire) levels. The importance of including map products that are representative of field conditions at the anticipated time of ground visits to proposed seeding sites was reinforced by the SNC experience.

See Chapter 9 in the full report for more details on remote sensing assessments and application relevant to the SNC.

DISCUSSION

Ely District BLM staff learned a number of lessons in the process of creating and implementing the SNC ES and BAR Plans. These lessons are outlined below beginning with 1) lessons learned (how we would do things differently with regards to specific tasks), followed by 2) management considerations for future (landscape scale) fires, and 3) future

work needed. All of the lessons, management considerations and future work have the potential to improve planning and implementation effectiveness on future fires managed in the Ely District.

Lessons Learned

Identifying Seeding Polygons. Identifying seeding polygons on large-scale fires presents challenges, because a combination of GIS and ground-truthing must be used to practically assess seeding sites. In the case of the SNC, we used course-scale GIS data. While rather extensive ground reconnaissance of the burned areas was conducted, individual areas proposed for treatment could not be thoroughly evaluated, given time constraints for implementing treatments in seasonally effective windows. As a result, a small portion of the mesic blackbrush seeding treatment dipped into potential desert tortoise habitat. This resulted in non-native species inadvertently being seeded into desert tortoise habitat.

In addition, based on ground-truthing, DOQQs (aerial photos) were found to be the most accurate of a variety of GIS and remote sensing-derived datasets in delineating the presence of trees in PJ woodlands. However, this method was not so good at distinguishing interior chaparral communities from pinyon-juniper woodlands and meant some interior chaparral areas that were not intended for seeding (because they will regenerate naturally) were seeded. Burned mesic blackbrush communities were more difficult than PJ woodlands to delineate using GIS. Ground reconnaissance was therefore crucial in differentiating burned blackbrush from other burned vegetation types.

Scientists and land managers working on the SNC fires are also realizing that BARC-derived indices of burn severity and vegetation mortality, commonly used in post-fire resource assessments, do not always work well for shrub-dominated desert scrub ecosystems. This is especially true with blackbrush (*Coleogyne ramosissima*)-dominated communities. Because these shrub communities tend to have very low vegetation biomass pre-fire, overall biomass consumption is low. BARC imagery correlates low consumption with low vegetation mortality. Blackbrush is extremely flammable, however, and is usually completely consumed by even low severity fires. This was the case in the SNC. While most of the burned blackbrush communities on the SNC were rated as low vegetation mortality by BARC-derived indices, these areas actually had nearly 100% vegetation mortality and a very high ecological burn severity. This further highlights the importance of ground reconnaissance in blackbrush communities and perhaps further work refining BARC-derived indices in ecosystems with low vegetation biomass.

Unburned Reference Plots. The SNC contained a number of unburned plots. These plots were only monitored in year 1 post-fire. They provided valuable insight into what vegetation structure and species composition may have been like in the burned areas if the

SNC fires had not occurred. In the future, unburned reference plots should be monitored as long as vegetation recovery is monitored, if at all possible, to allow succession trajectories in the burned areas to be compared to those in unburned areas.

Seedling Planting. A number of seedlings were grown out for planting on the SNC. A large number of the seedlings molded during transport and storage, because we were unaware that the bare root seedlings needed to remain in cool storage from the time they were picked up to the time they were planted. The blackbrush seedlings fared particularly poorly from this treatment while fourwing saltbush and spiny hopsage fared better during transport and storage. The fourwing and spiny hopsage seedlings did extremely well after planting until grasshoppers defoliated the spiny hopsage plants. It is unknown how this ultimately affected spiny hopsage seedling survival. Thirty-two blackbrush seedlings survived and continue to grow. The blackbrush seedlings that survived were healthy when they arrived, suggesting that blackbrush seedling survival can be improved if transport and storage methods are improved. Lastly, of six treatments tested for improving seedling available moisture, tree shelters proved to be the most successful aid to blackbrush establishment on the SNC. We recommend their continued use in future seeding plantings.

Improving Seeding Success and Post-Fire Vegetation Recovery. The SNC fires destroyed many fences used to control livestock. Unfortunately, less than half of the originally planned fence repairs were completed due to funding, staff workload and contracting limitations. As a consequence, cattle were reported on the burned areas of at least 14 of the 27 allotments on the Ely BLM managed SNC fires between 2006 and 2008, including within mesic blackbrush seeding polygons. Feral cattle further complicated the situation. There is a need to improve implementation and enforcement of fencing treatments or reassess implementing unprotected seedings in the future.

Staffing Issues. A number of issues hampered the implementation of the SNC ES and BAR Plans. The 2005 fire season was very busy with five other fires to stabilize in the Ely District in addition to the SNC fires. 2006 was also busy with 44 fires requiring stabilization. Staff workloads were overwhelming. The ES plan proposed funding a dedicated Implementation Coordinator. This position was not funded. Funding this position and filling it in a timely manner could have improved the successful implementation and administration of the SNC ES and BAR treatments. In addition, the ESR program has no base funding. This leads to continual staff turnover. The loss of experienced staff presents a larger issue in the program that needs to be addressed.

Scale Issues. The scale and remote nature of the SNC led to all sorts of logistical issues that affected the successful stabilization and rehabilitation of these fires. Scale affects the ability to effectively assess resource impacts, plan and implement treatments and monitor

and manage treatments. It affects physically covering distances with regards to implementing treatments, particularly ground-based treatments. This exacerbates time, staff and funding constraints. In addition, the cost of effectively stabilizing and rehabilitating fires of this magnitude means making tough choices and prioritizing treatments and treatment areas. Landscape scale fires also increase the manpower needed to implement treatments and the workload of a number of different programs including contracting. Contracting can cause significant hurdles to treatment implementation on projects of this scale, particularly when you have a number of large contracts all requiring quick turnaround. The situation is further exacerbated by the timing of fires in the Great Basin and Mojave that often occur near or after contracting deadlines for a given fiscal year.

Monitoring Issues. The scale and remote nature of the SNC also caused issues with monitoring protocols. Due to time and funding constraints, some of the sampling methodologies developed for the SNC had to be modified over the three-year sampling period. These changes (particularly the change from line-point intercept to ocular cover estimates) may have weakened the strength of some of our analyses and certainly caused conflict between the USGS (who was analyzing the BLM data) and ESR program staff. This conflict generally surrounded a lack of understanding on the part of the USGS as to: 1) what it takes to implement intensive monitoring protocols in terms of time, people and dollars; and 2) the need for monitoring which allows land managers to answer landscape scale management questions. Landscape scale fires highlight the need for balance between the ideal and the realistic approach to monitoring. They also highlight the need for a review of current ESR and BLM monitoring policy objectives. The reality of implementing intensive and effective monitoring over large areas, for example, requires considerable human and financial resources and, therefore, a commitment by policy managers to significantly enhanced funding for monitoring.

Policy managers also need to look at extending the three year ESR monitoring period. Land managers typically believe that seeding treatments can take more than three growing seasons to establish. Therefore, three years is probably not long enough to detect the establishment of adult plants from seedings or temporal trends in establishment of seeded species/vegetation recovery particularly in arid lands. Serious consideration should be given to adopting a design similar to that used by the National Park Service for fire effects monitoring collected at intervals of 1, 2, 3, 5, 7, 10, and 15 years post-fire (USDI National Park Service 2001).

Management Considerations for Future Fires

Improving Seeding Success and Post-Fire Vegetation Recovery. USGS findings suggest that seeding communities in the Mojave that are intolerant of fire with a wider variety of woody species could increase rates of woody cover recovery and, potentially,

reduce abundance of non-native annual grasses and forbs over time. This is worth consideration for future fires.

Another modification to seeding treatments worth exploring is the pattern and extent of seeding. The USGS suggests that seeding many smaller areas, instead of blanket seeding larger areas, could potentially enhance seeding success. The smaller seeded areas would create vegetated islands that, over time, serve as colonizing sources of seeded species into the surrounding landscape. In addition, given cheatgrass, red brome, and *Erodium* abundance appears to vary across a precipitation gradient, one or more of these species may dominate in all years. Therefore, yet another approach to seeding may be to target specific vegetation types (or specific elevation zones) in years with a particular rainfall pattern (e.g. seed higher elevation areas in dry years but not wet years), and/or limit seeding to when field examinations show low populations of annual grasses or forbs. This would increase treatment implementation costs, but it could improve treatment success and reduce ecological costs.

Collecting seed bank data in burned and unburned areas in future projects could also be a relatively easy and inexpensive way of quantifying if, for example, non-native annuals were significant components of the aboveground vegetation, the seed bank, or both pre-fire and help in evaluating seeding sites, post-fire succession patterns and effectiveness of seeding treatments.

In addition, rather than evaluating aerial seeding as successful or unsuccessful based on pre-determined desired densities of established plants (the current approach), a new approach that identifies and evaluates the conditions under which post-fire aerial seeding would have the greatest likelihood of success could be very beneficial.

Application of annual grass-specific herbicides prior to seedings might also help to reduce competition of the non-native annual grasses with seeded species. It is something to consider in future BAR treatments and smaller ES treatments where categorical exclusions may allow practical implementation of herbicide and seeding treatments in the one year treatment window.

Future Work Needed

Improving Seeding Effectiveness. Results of post fire seeding in desert tortoise habitat on the SNC suggest that broadcast seeding has strong potential to provide herbaceous plants for forage and long-term perennial plant cover to support tortoise recovery in burned habitats. Maximizing seeding rates, focusing on a combination of native species that can withstand disturbance conditions (including species that are found in adjacent unburned areas), and reducing seed residence time with seasonally-appropriate application may

improve establishment of native annuals and fast-establishing perennials more generally post-fire and deserves further research.

In addition, more information is needed on how seeded species establishment rates vary with the seasonal timing of seeding, species composition of seed mixes, application rates, and among vegetation types and years of contrasting climatic conditions. Establishment of small scale field and/or greenhouse competition and resource availability experiments would allow us to better evaluate not just what the post-fire succession patterns are in the Ely District, but the relative importance of different mechanisms producing the patterns, and should be considered.

Sixteen species were used in the aerial seed mixes applied to the SNC fires. Of these, several species appeared to have a much higher chance of establishment in the first three years in at least one of the vegetation types evaluated. The best establishing native species were Indian ricegrass, bottlebrush squirreltail, galleta grass and sandberg bluegrass. The best establishing non-native species were crested wheatgrass and small burnet. These species should be considered as candidates for systematic evaluations of their likelihood of establishment in the Ely district in the future.

Improving Treatment Effectiveness Monitoring. Remote sensing may provide a lower cost method for monitoring vegetation trends in burned areas that are seeded versus those that are not seeded. However, additional study is required to assess whether the resolution and frequency of remote sensing coverage is sufficient to detect vegetation trends that may occur among burned and unburned areas, and areas where different management treatments are applied.

CONCLUSION

The SNC was a significant ecological event for the Mojave Desert and continues to present many land management challenges in southern Nevada and for the Ely District BLM. In the past 25 years, invasive non-native exotics such as cheatgrass and red brome have invaded the desert southwest providing fuels that carry fire in areas that historically did not carry fire. This trend is expected to continue. Lessons learned on the SNC and recommendations for future actions will hopefully improve future approaches to landscape scale ESR efforts.

Complete chapters and additional materials, including appendices, maps, tables, figures and raw monitoring data can be found in the complete SNCESBAR report, available on the Ely District BLM internal website or on CD.

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AA – Additional Aerial Seeding Coverage Plot
ACEC – Area of Critical Environmental Concern
AFB – Air Force Base
AIC – Akaike's Information Criterion
AICc - Akaike's Information Corrected Criterion
ANOVA – Analysis of Variance
ATV – All-Terrain Vehicle
AUM – Animal Unit Month
AVHRR - Advanced Very High Resolution Radiometer
AWiFS - Advanced Wide Field Sensor
BAER – Burned Area Emergency Response
BAR - Burned Area Rehabilitation
BARC - Burned Area Reflectance Classification
BB – Brushbelt
BLM – Bureau of Land Management
BRD – USGS Biological Resources Discipline
BRRC – *Bromus madritensis* (red brome) Cover
BRTC – *Bromus tectorum* (cheatgrass) Cover
CART – Classification and Regression Tree
CBI – Composite Burn Index
CCA – Canonical Correspondence Analysis
DEM – Digital Elevation Model
dNBR – Differenced Normalized Burn Ratio
dNDVI - Differenced Normalized Difference Vegetation Index
DOQQ - Digital Orthophoto Quarter Quadrangle
DM - Demonstration
DRG – Digital Raster Graphic
DWMA - Desert Wildlife Management Area
EAGEC - Exotic Annual Grasses and *Erodium cicutarium*
EAGS - Exotic Annual Grasses and Shrubs
EANA - Exotic Annuals and Native Annuals.
ENLC – Eastern Nevada Landscape Coalition
EROC – *Erodium cicutarium* (storksbill) Cover
EROS – USGS Earth Resources Observation and Science Center
ES – Emergency Stabilization
ESR, ES&R – Emergency Stabilization and Rehabilitation
FY – Fiscal Year
GBI – Great Basin Institute
GIS – Geographic Information Systems
GLM – Generalized Linear Model
GPS – Global Positioning System
HMA – Herd Management Area
IAGC – Invasive Annual Grass Cover
IAGD – Invasive Annual Grass Density

IGO – Intra-Governmental Order
JFS – Joint Fire Science
KS Test - Kolmogorov-Smirnov Test
LCM – Livestock Compliance Monitor
MBB – Mesic Blackbrush
MEAP - Mixed Exotic Annuals and Perennials
MFT – Multiway Frequency Tables
MLRA – Major Land Resource Area
MODIS - Moderate Resolution Imaging Spectroradiometer
MP - Mixed Perennials
MRLC – Multi-Resolution Land Characteristics
MSL – Mean Sea Level
MTBS – Monitoring Trends in Burn Severity
NAIP - National Agriculture Imagery Program
NAP - Native Annuals and Perennials
NBR – Normalized Burn Ratio
NCC – Nevada Conservation Corps
NDOW – Nevada Department of Wildlife
NDVI - Normalized Difference Vegetation Index
NIR – Near Infrared
NOAA – National Oceanic and Atmospheric Administration
NPS – National Park Service
NR – Natural Regeneration or National Register
NWPJ – Non-Wilderness Pinyon-Juniper
NWS – National Weather Service
OHV – Off Highway Vehicle
PJ – Pinyon-Juniper
PRNC – Perennial Cover
PRND – Perennial Density
QA – Qualitative Assessment
RdNBR – Relative Differenced Normalized Burn Ratio
ReGAP – Regional Gap Analysis Project
RS – Remote Sensing
RSAC – USFS Remote Sensing Applications Center
SCA – Student Conservation Association
SHRC – Shrub Cover
SLC – Scan Line Corrector
SNC – Southern Nevada Complex
SNPLMA – Southern Nevada Public Land Management Act
SOW – Statement of Work
SWIR – Short Wave Infrared
T&E – Threatened and Endangered
TM – Landsat Thematic Mapper
TVGC – Total Vegetation Cover
USFS – United States Forest Service
USFWS – United States Fish and Wildlife Service

USGS – United States Geological Survey
USDA – United States Department of Agriculture
USDI – United States Department of the Interior
VIF – Variance Inflation Factors
VNIR – Visible/Infrared
WERC – USGS Western Ecological Research Center
WO – BLM Washington Office
WPJ – Wilderness Pinyon-Juniper

Chapter 1: Introduction

BACKGROUND

Between June 22, 2005 and July 10, 2005, the 11 fires that make up the Southern Nevada Complex (SNC) burned 739,037 acres in southeastern Nevada, southwestern Utah and northwestern Arizona. The Ely District Bureau of Land Management (BLM) manages 597,096 acres across four of these fires (Table 1-1). The location and spatial extent of the Ely BLM burned areas are displayed in Figure 1-1. Smaller portions of the burned areas are managed by the Las Vegas District BLM, US Fish & Wildlife Service, Forest Service, Bureau of Reclamation, and National Park Service. These areas are not included in this report.

Table 1-1. Summary of Ely District BLM acres burned.

Fire	Ely BLM Acres Burned
Delamar	168,007
Duzak	214,038
Halfway	66,487
Meadow Valley	148,564
Total	597,096

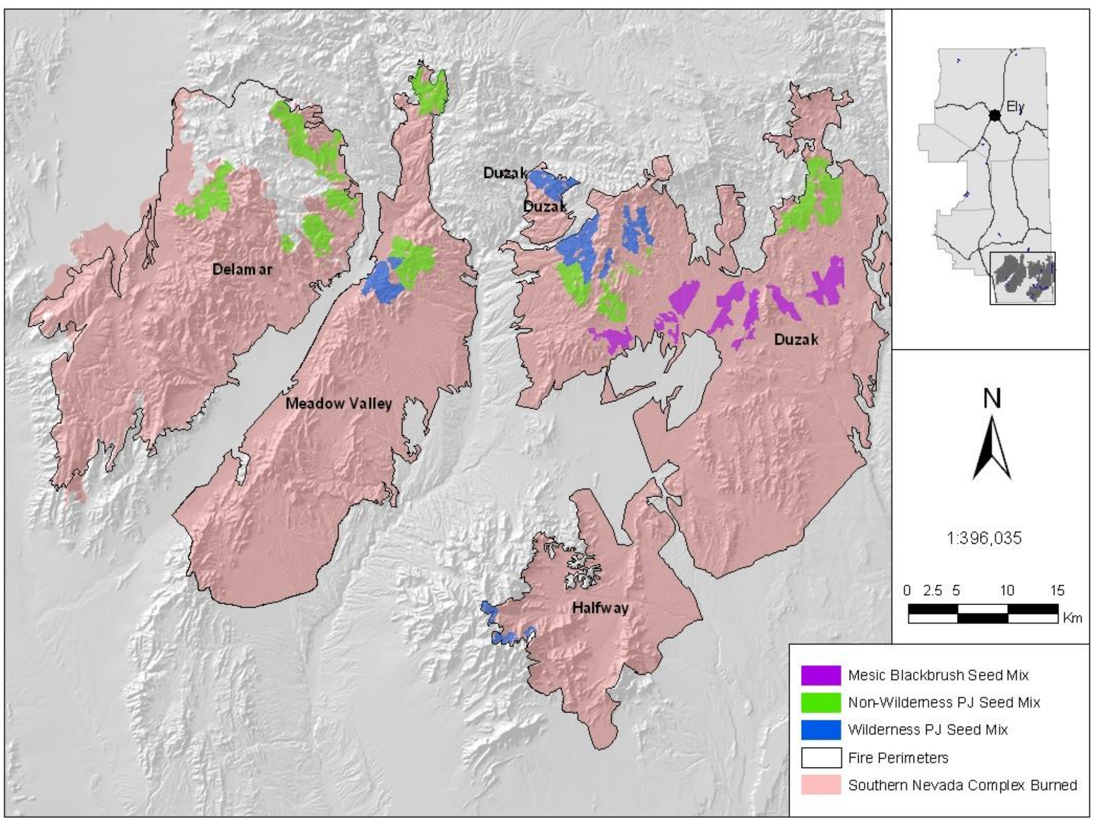

Figure 1-1. Location and spatial extent of the Southern Nevada Complex fires.

The SNC fires were initiated on June 22, 2005 by dry lightning storms and spread quickly due to high fuel loads and winds. Heavy rains during the previous winter and spring (resulting in more than 400% of normal rainfall from January through April 2005) resulted in unusually high production of non-native annual grasses (predominantly red brome and cheatgrass) that were able to carry fire through areas of normally sparse vegetation. These fine fuels served as ladder fuels between grasses, forbs, shrubs and trees of at least 12 vegetation communities in the Mojave Desert and Great Basin regions.

The SNC fires exhibited extreme fire behavior, and initial fire starts grew rapidly. Many times starts had grown to 1000 acres or more before initial attack. The situation was complicated by restrictions to suppression activities as a result of environmental concerns about endangered desert tortoise habitat. Initially this limited the use of engines and dozers to existing roads. The most common suppression tactics were retardant and water drops, backfires, and using roads and geographic features. Hand line construction was limited by fast moving fire and high rates of spread. (See Appendix 1-1 for additional suppression information.)

GEOGRAPHIC SETTING

Physiography

The SNC fires burned primarily in the Basin and Range Physiographic Province. The fires exhibit typical basin and range topography: relatively narrow north-south-tending mountain ranges separated by wider alluvium-filled basins. SNC elevations range from about 2,000 feet in the southeastern valleys to over 8,500 feet in the North Pahranagat Range. More specifically, they range from approximately 3500 feet to 7800 feet MSL (mean sea level) for the Delamar Fire, 3900 feet to 7600 feet MSL for the Duzak Fire, 2450 feet to 7350 feet MSL for the Halfway Fire, and 2500 feet to 7000 feet MSL for the Meadow Valley Fire. The crests of the major ranges are generally 3,000 feet to 4,000 feet above the adjacent basins, with elevations of both the mountains and basins generally lower in the southern half of the burned area and increasing to the north.

The SNC was a large scale fire. It burned an area roughly 50 miles wide and 40 miles long. Large expanses of the Delemar, Meadow Valley, Clover, and Mormon mountain ranges were burned, as were the valleys between these ranges. The burned area terrain is highly variable and encompasses a wide variety of slopes, elevations, vegetation and soil types.

Average precipitation throughout the burned area is low, and most surface runoff infiltrates into the soils before it reaches the basin floors. Major flooding is caused by winter storms with low intensity rainfall over wide areas, often continuing for several days, or rain-on-snow events in the higher elevations. Flash flooding can occur in any area at any time of

year but is most probable during the summer and fall. Surface water resources in the area are limited and groundwater is scarce.

Geology and Soils

General geology of the SNC burned region consists of Quaternary unconsolidated alluvial, colluvial, eolian, and playa deposits in the basins covering older units. Older units include Tertiary bimodal volcanic and sedimentary units covering much older Precambrian to Paleozoic igneous volcanic, intrusive, sedimentary, and metamorphic units. Tertiary to Quaternary faulting, uplift and erosion have exposed these units throughout the region.

The dominant soil orders found within the burned areas are Aridisols, Entisols, and Mollisols. These mineral soils are low in organic matter with layers that are highly variable in thickness, texture, rock fragment content, and physical and chemical properties. Elevation, geology, climate, vegetation, and landform have a strong influence on the distribution of the soils in the region.

The susceptibility to erosion for a soil varies with geology, parent material, elevation, slope, aspect, vegetative cover, microclimate, land use, and landscape history. Wildfire consumption of protective plant and litter cover leaves soils highly susceptible to water and wind erosion. The following landforms represent the major types found on the burned areas (Table 1-2).

Table 1-2. Major landforms found in the Ely BLM burned areas.

Landform	Water Erosion Hazard	Wind Erosion Hazard
Sand Sheet	Slight	High
Fan Piedmont	Slight to moderate	Slight
Mountains	Moderate to high	Slight

The relative degrees of erosion potential by wind or water are generally inversely related. Much of the Mojave Desert consists of naturally sparse vegetation communities that provide little vegetative cover, even in an unburned condition. In these areas, there is likely to be little increase in erosion susceptibility due to a fire. Further, many of these areas are characterized by surface erosion pavement (rock fragments that have concentrated on the surface over time as past cycles of wind and water erosion have occurred). Where this surface armor occurs, susceptibility to erosion by wind and water is naturally low, both before and after a fire.

Climate

The SNC experiences an arid to semi-arid climate, with an average annual rainfall of 4-20 inches depending on elevation. Summers tend to be hot, dry and windy, with highest temperatures (exceeding 100 degrees Fahrenheit) recorded during July and August. Freezing temperatures often occurs during the winter, particularly in higher elevation regions. Average

annual precipitation is highly variable from one year to the next. Annual precipitation averages less than 4 inches in the lower elevation desert scrub areas, 15-18 inches in mid-elevation chaparral communities, and more than 20 inches in forest stands on upper slopes. Almost all precipitation arrives in the winter months of January, February and March, but the region also experiences rare, intense summer thunderstorms during the monsoon season. (See Chapter 9 for more detailed precipitation data.)

Lightning events are common in the Basin and Range Province and increase in frequency with elevation. The majority of lightning occurs in July and into August during the monsoon season and is a primary source of wildfire ignition. In the past 25 years, invasive non-native exotics such as cheatgrass and red brome have invaded the desert southwest providing fuels that carry fire in areas that historically did not carry fire. As a result, larger, more frequent fires can occur, particularly after wet winters, as exemplified by the SNC.

Hydrology and Water Quality

Most stream channels within the SNC are ephemeral, flowing only during and immediately after rainfall events. Stream flows are very flashy with sudden increases and decreases in flow. Debris, including vegetation, sand, rocks, and large boulders, are transported downstream during these storm events. Significant flood events, such as the one triggered by a January 2005 rain-on-snow precipitation event, can produce dramatic changes to the desert landscape.

The Meadow Valley Wash is the only perennial stream within the SNC boundaries. Floods resulting from the January 2005 rain-on-snow event rearranged the majority of its channel. Floods from the same event also rearranged the majority of both the Beaver Dam Wash and Virgin River channels. The SNC fires burned watersheds that drain into both Beaver Dam Wash and the Virgin River. Small springs (generally flowing less than 5 gallons per minute) and seeps provide isolated and limited water for plants, wildlife, or domestic purposes on the SNC. Many of these springs have been altered by the installation of retention dams, pipelines, and troughs for livestock use.

Air Quality

Barren rock, alluvium, and dry lakebeds are all sources of dust and sand in the Mojave Region. Processes involving sand and dust transport play an important roll in shaping the landscape and the ecosystem of the Mojave. Typically most dust (clay and silt), including valuable topsoil, becomes suspended in the wind and is carried away from the region by prevailing winds. However, dust that settles into stony soils of the desert provides improved moisture retention and adds nutrients. Sand, in contrast, is moved along the surface by wind as a saltating bedload. High dust concentrations in the air can create near “white-out” conditions.

Vegetation

The Ely BLM SNC fires occurred within two Major Land Resource Areas (MLRAs). The southern end of the fires occurred within the Mojave Desert of the Sonoran Basin and Range MRLA, while the northern end of these fires occurred within the Southern Nevada Basin and Range MRLA (NRCS 2002 referenced in: USDI National Interagency BAER Team 2005). A variety of vegetation communities occur within the boundaries of the fires, including Creosote-White Bursage, Mixed Desert Scrub, Mojave Mid-Elevation Mixed Desert Scrub, Mixed Salt Desert Scrub, Interior Chaparral, Pinyon-Juniper Woodlands, Sagebrush-Dominated, and Montane Conifer Forests (generally Ponderosa pine stands) (Figure 1-2). The distribution of plant communities across the SNC is greatly influenced by rainfall and topography.

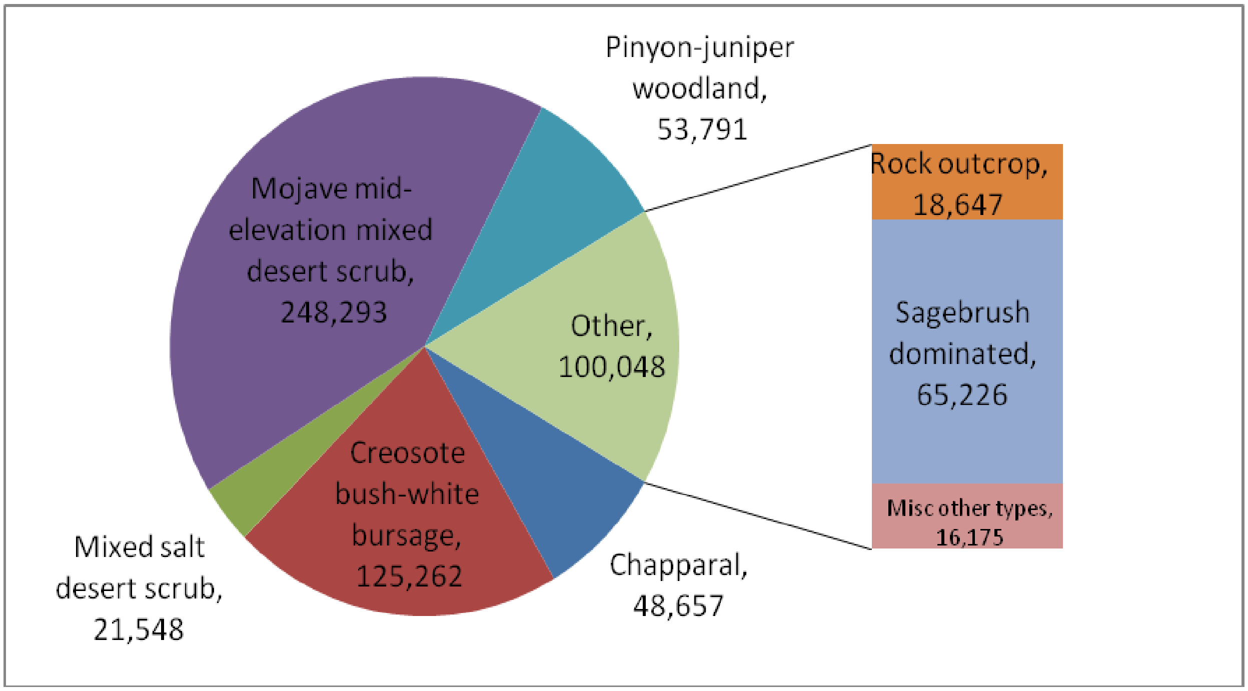

Figure 1-2. Vegetation types burned by the SNC fires in the Ely District.

BAER TEAM CALLOUT

Prior to the SNC, the largest fire the Ely BLM District had experienced was the 1999 Delemar Fire which burned 22,592 acres. Due to the size and scope of the fire, and BLM staff limitations both locally and regionally in particular, a national Burned Area Emergency Response (BAER) team was called in to assist with the initial assessment of SNC fire damage to BLM lands and development of the Emergency Stabilization (ES) plan.

The primary objectives of the BAER process and an ES plan are to:

- Prescribe post-fire mitigation measures necessary to protect human life, property, and critical cultural and natural resources; and
- Promptly mitigate the unacceptable effects of the fire on lands within and adjacent to the burned area in accordance with Department of the Interior (DOI) and BLM management policy guidelines, land management planning documents, and all relevant federal regulations.

The BAER team received an initial briefing on July 13, 2005 and commenced with resource assessments to determine the effects of the fire on soil and watershed, vegetation, wildlife, cultural, and recreation resources throughout the area, using aerial and ground reconnaissance methods. They worked with local resource advisors, BLM program staff, landowners, permittees, and researchers to assess ES issues, recommend ES treatments and document longer-term rehabilitation needs. Potential values at risk identified for the SNC included: homes, buildings, and other structures; federal and state listed species and Sensitive species; roads, railroad lines, and power lines; and cultural resources. Resource assessment findings and recommendations are detailed below.

RESOURCE ASSESSMENT FINDINGS & RECOMMENDATIONS

Soils and Watershed

BAER team soil and watershed specialists conducted a rapid assessment of threats to life, property, and critical natural resources within and downstream of the SNC fires. They used aerial and on-the-ground reconnaissance to determine risk of increased runoff, erosion or dust from the fires.

Most of the burned area experienced low soil burn severity with some areas of moderate to high severity (Table 1-3, Figure 1-3). The predominately light loads of fine fuels, such as grass, resulted in short fire residence times and consequently lower soil burn severity. Areas of moderate soil burn severity tended to occur where pre-fire vegetation consisted of shrubs or less dense pinyon-juniper, and the relatively small areas of high soil burn severity occurred almost entirely at upper elevations in the mountains where ponderosa pine or pinyon-juniper communities had grown relatively dense.

Table 1-3. SNC fire soil burn severity.

FIRE NAME	SOIL BURN SEVERITY	ACRES	PERCENT
Delamar	1-Unburned to Very Low	54,234.0	32.3
	2-Low	87,136.4	51.9
	3-Moderate	25,710.6	15.3
	4-High	962.1	0.6
Delamar Total		**168,043.1**	**100.0**
Duzak	1-Unburned to Very Low	76,521.5	32.2
	2-Low	141,405.2	59.5
	3-Moderate	19,525.3	8.2
	4-High	110.5	0.0
Duzak Total		**237,562.6**	**100.0**
Halfway	1-Unburned to Very Low	18,253.3	27.4
	2-Low	47,861.0	71.9
	3-Moderate	461.4	0.7
Halfway Total		**66,575.7**	**100.0**
Meadow Valley	1-Unburned to Very Low	27,615.9	18.5
	2-Low	107,865.1	72.4
	3-Moderate	13,341.1	9.0
	4-High	65.8	0.0
Meadow Valley Total		**148,888.0**	**100.0**

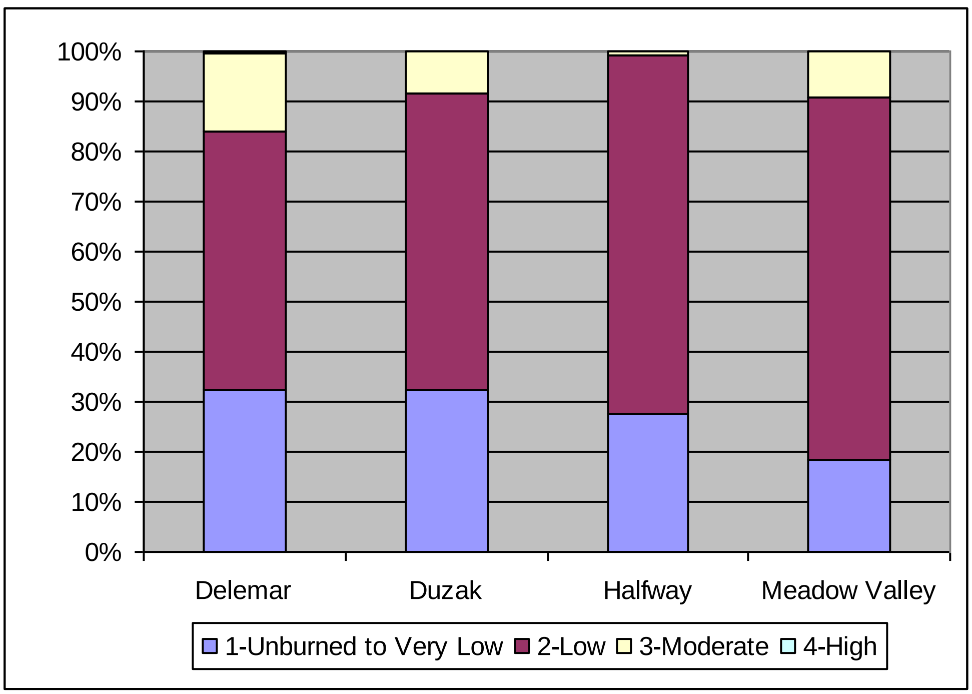

Figure 1-3. SNC Complex soil burn severity graph.

In the Mojave Desert ecosystems, pre-fire vegetation was generally sparse, with invasive grasses mostly concentrated underneath the shrubs. In these areas, soil burn severity was very low to low, but with high vegetation mortality. Shrub communities were completely

consumed. In the higher elevations of the mountainous areas where pinyon-juniper communities had grown dense, soil burn severity was high with high vegetation mortality.

Soil burn severity relates specifically to effects of the fire on soil conditions (e.g., amount of surface litter and duff, infiltration rate, erodibility, soil structure). In areas where pre-fire vegetation was sparse or consisted of light, “flashy” fuels such as grass, and where heat residence time was very short, complete consumption of vegetation by fire may have occurred with little effect on soil properties. Increased residence time can promote the formation of water repellent layers at or near the soil surface, loss of soil organic matter, and loss of soil structural stability, resulting in increased runoff and soil particle detachment by water and wind, and transport off-site.

On the Delamar Fire, areas of high soil burn severity were found to occur on soils with high to moderate soil erosion hazard. The high and moderate soil erosion hazard on the SNC fires (Table 1-4) can be attributed to the steepness of the terrain and relative amount of surface fines exposed.

Table 1-4. SNC soil erosion hazard.

Fire Name	Low Soil Erosion Hazard (% of burned area)	Moderate Soil Erosion Hazard (% of burned area)	High Soil Erosion Hazard (% of burned area)
Delamar	50 %	40 %	10 %
Duzak	> 73 %	25 %	< 2 %
Halfway	70 %	15 %	15 %
Meadow Valley	90 %	10 %	0 %

The terrain on the SNC fires is steep mountainous to rolling hills with intermittent to ephemeral channels. Channels drain west to Meadow Valley Wash and east to Beaver Dam Wash on the Duzak Fire, west to Meadow Valley Wash and southeast to the Virgin River on the Halfway Fire, and east to Meadow Valley Wash and west to Kane Springs Wash on the Meadow Valley Fire.

During intense storm events post-fire, roads on federal lands in the SNC were considered to be at a slight increased risk of flooding or being inundated with dead vegetation, sediment, and rock. The BAER team recommended posting warning signs to inform the public to stay out of the SNC area during rain events and for up to one hour following rainfall to mitigate public safety risks.

No values at risk were identified within burned areas of the Duzak and Meadow Valley fires. However, many potential risks to life, property, and natural resources exist downstream, including: a major trans-continental Union Pacific Railroad track following the drainage in Meadow Valley Wash; the communities of Moapa and Glendale; and the BLM

Sensitive species Meadow Valley Wash speckled dace and desert sucker. For the Duzak Fire additional values at risk include: high-voltage transmission lines and a dam along Beaver Dam Wash; the communities of Beaver Dam, Mesquite, and Bunkerville; and the BLM Sensitive species Virgin River spinedace in Beaver Dam Wash, and in the Virgin River the Threatened and Endangered (T&E) woundfin, Virgin River chub, southwestern willow flycatcher, and Candidate species yellow-billed cuckoo. The BAER team therefore recommended further assessments of Meadow Valley Wash (for the Meadow Valley and Duzak fires) and Beaver Dam Wash (for the Duzak Fire) to fully assess threats to values at risk and to develop potential treatments if deemed necessary.

Fire effects identified for the Halfway Fire were minimal. Scorched shrubs and blackened litter were discontinuous and soil burn severity was predominantly low. Fire effects were not expected to increase normal processes of erosion or runoff, and no watershed treatments were recommended. No downstream values at risk were identified for the Delamar Fire, and no watershed treatments were recommended.

Vegetation

BAER team vegetation specialists assessed vegetation resources for post-fire mortality or degree of top kill, potential for vegetative recovery, impacts from fire suppression activities, presence and potential for spread of non-native undesirable annuals or noxious weed species, and fire effects to T&E species. Vegetation groups within the burns were classified based on Burned Area Reflectance Classification (BARC) imagery into four mortality levels: unburned to low, with vegetation mortality of 0 to 5%; low to moderate, with a range of mortality of 6 to 60% (due to the extreme rates of spread and fire behavior); moderate, with 61 to 85% mortality; and high, with 86 to 100% mortality (Table 1-5). (Vegetation mortality, for the purposes of this BAER assessment, refers to immediate post-fire mortality of the above-soil plant parts and does not imply that the vegetation could not releaf or resprout from root crowns or epicormic plant parts.)

A majority of the burned area experienced low to moderate vegetation mortality (71%) with some areas of unburned to very low (15%), moderate (13%), or high mortality (1%). For the Mojave and Great Basin vegetation types, low to moderate was incorporated into higher mortality, since on-site visits indicated soil reflectance in the low to moderate reflectance category had high vegetation mortality in certain vegetation types. It is likely that actual vegetation mortality is still underrepresented. The majority of the high vegetation mortality class fell in those plant communities of the Mojave Desert that are not adapted to fire. While fire intensity varied throughout the burned area, the rapid rate of fire spread resulted in consumption of most of the grasses and herbaceous species, and some of the shrubs and scattered trees.

Scientists and land managers working on the SNC fires are realizing that BARC-derived indices of burn severity and vegetation mortality do not always work well for shrub-dominated desert scrub ecosystems. This is especially true with blackbrush (*Coleogyne ramosissima*)-dominated communities. Approximately 300,000 acres of this vegetation type burned in the SNC fires. Because these shrub communities tend to have very low vegetation biomass pre-fire, overall biomass consumption is low. BARC imagery correlates low consumption with low vegetation mortality. Blackbrush is extremely flammable, however, and is usually completely consumed by even low severity fires. This was the case in the SNC. While most of the burned blackbrush communities on the SNC were rated as low vegetation mortality, these areas actually had nearly 100% vegetation mortality and a very high ecological burn severity.

While many species can resprout following fire, blackbrush is not a fire-tolerant species and generally never resprouts. Instead blackbrush relies upon seedling recruitment to maintain populations. It uses timing mechanisms that allow for seed germination only when certain favorable environmental conditions are met (Pendleton and Meyer 2004). These conditions are not likely to occur often. Studies indicate that blackbrush can take upwards of 60 years to reestablish (Anderson 2001, in: USDI Bureau of Land Management 2005) and possibly 1,000 years to be fully restored (Web et al. 2001, in: USDI Bureau of Land Management 2005). Other studies suggest that burned blackbrush sites are converted to other vegetation types and do not return as blackbrush sites (Callison et al. 1985 and Haines et al. 2003, in: USDI Bureau of Land Management 2005). Due to concern about long-term blackbrush loss, specialists at the Ely District wrote a Burned Area Rehabilitation (BAR) Plan in the fall of 2005 with a focus on reestablishing blackbrush.

Table 1-5. Vegetation mortality classes for the SNC fires.

Vegetation Mortality	Characteristics
Unburned to Very Low	Unburned areas and areas of very low vegetation mortality. Pre-fire vegetation was sparse or consisted of non-native grasses between widely spaced native species of shrubs, cacti, and trees.
Low to Moderate	Shrub canopy may be scorched or consumed. Grasses are consumed. Unburned patches between shrubs and trees.
Moderate	Shrub canopy consumed, with stobs or stems left. Pinyon and juniper or pine canopy consumed, with branches remaining. Unburned patches between shrubs are smaller but still present.
High	Dense stands of pine or pinyon-juniper at higher elevations, canopy completely consumed with few branches remaining.

Table 1-5. Continued.

FIRE NAME	MORTALITY CLASS	ACRES	PERCENT
Delamar	1-Unburned to Very Low	47,782.8	28.4
	2-Low to Moderate	82,483.0	49.1
	3-Moderate	36,832.1	21.9
	4-High	927.1	0.6
Delamar Total		**168,025.0**	**100.0**
Duzak	1-Unburned to Very Low	18,886.9	8.0
	2-Low to Moderate	191,239.6	80.5
	3-Moderate	27,271.3	11.5
	4-High	139.2	0.1
Duzak Total		**237,537.0**	**100.0**
Halfway	1-Unburned to Very Low	2,999.0	4.5
	2-Low to Moderate	62,730.4	94.2
	3-Moderate	839.5	1.3
Halfway Total		**66,568.9**	**100.0**
Meadow Valley	1-Unburned to Very Low	20,449.4	13.7
	2-Low to Moderate	105,281.8	70.7
	3-Moderate	23,070.0	15.5
	4-High	69.1	0.0
Meadow Valley Total		**148,870.3**	**100.0**

Across the SNC, the spread of noxious and non-native invasive species was anticipated, due to high vegetation mortality and the presence of non-native invasive species pre-fire. Many non-native invasive plants exist throughout the burned areas and Clark and Lincoln Counties, Nevada. No T&E species were expected to be impacted, except the desert tortoise within the creosote-bursage habitat type. In areas where the Threatened desert tortoise occurs, the most direct threats are from the bromes, filaree, Sahara mustard (*Brassica tournefortii*) and *Schismus* spp. These species are also problematic in native desert vegetation communities and often problematic in burned shrub, woodland, and forest communities. When habitat type conversion occurs in upland communities, dominance by these species usually occurs.

All four SNC fires had permitted livestock grazing. Many of the permitted Animal Unit Months (AUMs) on the 27 allotments were potentially impacted by the extent of the burns and existing rangeland projects (i.e. fencing and water sources). The BAER team recommended closing the allotments to livestock grazing for a two-year minimum in pinyon-juniper areas to allow for recovery and stabilization. In the Mojave Desert ecosystem, which is also habitat for the federally listed Threatened desert tortoise, the burned areas would remain closed until minimum vegetative conditions were returned. Livestock grazing is anticipated to be affected by the SNC fires for the long term, particularly in the Mojave Desert ecosystem, where low precipitation results in lengthy recovery time. Moreover, much of the burned area occurred in rangelands and pinyon-juniper woodlands. Pinyon-juniper

areas often support very little herbaceous understory resulting in a conversion to non-native annuals that offer little forage beyond a few weeks early in the spring.

The SNC burned four Herd Management Areas (HMAs) in the Ely District (Table 1-6). The BAER vegetation team recommended conducting a census and gather of 250 wild horses and burros within 5 HMAs in the Ely Field Office jurisdiction, to ensure timely recovery of HMAs in the fire and neighboring areas, protect re-seeded sites, assist recovery of critical desert tortoise habitat, and mitigate weed invasion.

Table 1-6. HMAs in the Ely District burned by the SNC fires.

Herd Management Area (HMA)	HMA Acreage	Fire Name(s)	Acres Burned in HMA	% of HMA Burned
Delamar Mtns. (Ely)	183,557	Delamar and Meadow Valley	77,439 11,024	48%
Meadow Valley (Ely)	147,688	Meadow Valley	74,726	79%
Blue Nose Peak (Ely)	132,222	Duzak	51,350	61%
Clover Mtns. (Ely)	262,496	Duzak and Meadow Valley	24,586 8,714	20%

Wildlife

BAER team wildlife specialists assessed potential effects to federally listed species and their habitats that occur within or downstream of the SNC from fire, suppression actions, and proposed ES activities. Seven federally listed species (T&E) and one Candidate species occur within the SNC or downstream receiving water bodies and riparian areas. The potential effects of the fire and suppression actions to these species are summarized in Table 1-7. In addition, one experimental (nonessential) population of California condors may occasionally forage over the eastern edge of the fire area. Numerous BLM Sensitive species (most notably desert bighorn sheep, Gila monster, Meadow Valley Wash speckled dace, and Meadow Valley Wash desert sucker) and one Utah State Sensitive species (Virgin River spinedace) also occur within the SNC or downstream receiving water bodies and riparian areas.

A finding of “not likely to jeopardize” was reached for the nonessential experimental population of California condor, given the feeding habits of the California condor and the limited scope and extent of the treatment specifications. The Meadow Valley and Duzak fires did compromise the integrity of two of three identified suitable (but unoccupied) refugia sites for Big Spring spinedace, due to short-term loss of riparian overstory and potential sediment delivery to these two sites.

The SNC fires also further diminished important mule deer and elk habitats, especially blackbrush and pinyon-juniper communities in mid- to upper elevations. Although deer and elk are not BLM Sensitive species, they were specifically brought forward as issues by the Nevada Department of Wildlife (NDOW). Their habitats are declining in amount and quality rangewide due to drought, fire, fragmentation, and other factors.

Table 1-7. Potential effects of the SNC on federally listed species.

Species	Federal Status	Determined Effect
Desert Tortoise	Threatened	May affect, likely to adversely affect. The proposed BAER treatments are expected to have beneficial effects on the desert tortoise. However, adverse effects are not discountable, given observed mortalities and extent of suppression activities in desert tortoise critical habitat.
Bald Eagle	Threatened	No effect based upon the timing of the fire and the highly mobile nature of this species.
Southwestern Willow Flycatcher	Endangered	No effect based on the lack of fire and suppression activities in flycatcher habitat and the location of treatments in relation to its habitat.
Yuma Clapper Rail	Endangered	No effect based on the distance of the fire, suppression activities, and proposed treatments from the Yuma clapper rail and its habitat.
Virgin River Chub	Endangered	No effect due to the distance between treatment areas and its critical habitat.
Woundfin	Endangered	No effect due to the distance between treatment areas and its critical habitat.
Big Spring Spinedace	Threatened	No effect due to location of the fires, suppression activities and treatments in relations to its designated critical habitat.
Yellow-billed Cuckoo	Candidate	No effect based on the lack of fire and suppression activities in cuckoo habitat and the location of treatments in relation to its habitat.

Of the federally listed species that occur within or downstream of the SNC fires, only the desert tortoise is likely to be adversely impacted by the SNC fires. There were approximately 403,644 acres of potentially suitable habitat (less than 1,280 meters in elevation), and 32,682 acres of designated critical habitat for desert tortoise within the SNC fire area. (Not all of this acreage is managed by the Ely District BLM.) Desert tortoise mortalities as a direct result of the SNC fires were recorded by BAER team, US Fish & Wildlife Service and BLM staff. In long-lived species with low reproductive capacity, such as desert tortoises, losses of individuals can lead to population-level effects (Hailey 2000 and Esque et al. 2003, in: USDI National Interagency BAER Team 2005). Moreover, indirect effects of fire on desert tortoise may include permanent loss of habitat due to vegetation type-conversion or other ecosystem shifts (Brooks 1999, in: USDI National Interagency BAER Team 2005; Brooks and Esque 2002); abandonment of habitat in the burned area; decreased reproductive rates due to insufficient food, water, or shelter; and/or nutritional deficiencies due to a loss of food plants.

Two ES treatments were identified by the BAER wildlife team: 1) aerially seed two essential habitat components for desert tortoise with native shrubs (for cover) and forbs (for forage) on 4,945 acres of burned desert tortoise critical habitat; and 2) apply herbicide along 113 miles of road (3,270 acres) within the SNC to inhibit germination of red brome and cheatgrass. Earlier specifications for aerial seeding on a total of 47,000 acres of burned

pinyon-juniper and mesic brush habitat were expected to foster post-fire recovery of deer and elk winter and summer range as well. Two Rehabilitation (R) treatments were also identified by the BAER team: 1) repair two damaged desert bighorn sheep guzzlers; and 2) seed additional burned desert tortoise habitat areas identified by the Ely BLM staff.

Cultural Resources

There are 148 cultural resource sites located within the SNC lands managed by the Ely BLM. These lands have a varied prehistoric and historic cultural history. Aerial reconnaissance was completed for all the fires with rock art sites, rock shelters, buildings and structures, and mining sites. As possible, these sites were inspected for signs of fire intrusion. In addition, higher severity burn locations were inspected to gain a sense of the general impact and potential for indirect effects on archeological resources as a result of fires. A review of records and field inspection of seven potentially impacted sites indicated no sites were impacted by the fire or suppression activities and none appeared to be threatened by erosion. However, site looting and off-highway vehicle (OHV) damage were a concern.

Three ES treatments were identified by the BAER cultural resource team for the Ely BLM: 1) complete known cultural resource assessments at 34 sites to assess damage/loss and post-fire risks; 2) conduct law enforcement patrols of sites and close areas/roads until collector interest drops off and vegetation becomes re-established; and 3) consult with federally recognized tribes that have cultural ties to the area.

Operations

BAER team operations specialists conducted aerial reconnaissance and ground surveys to support resource teams and assess the degree of resource impacts resulting from the SNC fires. An increased chance of public exposure to potential hazards, including but not limited to abandoned mines, hazardous materials, road washouts, and public safety sign damage, and an increased workload associated with implementing the SNC ES Plan were identified.

Two ES treatments were recommended by the BAER operations team: 1) conduct field surveys of burned areas to identify public safety hazards; and 2) provide for an Implementation Leader to coordinate all aspects of implementing, tracking and reporting on ES actions approved in the SNC ES Plan.

Recreation

BAER team recreation specialists assessed the effects of fire and suppression activities on recreational opportunities. No developed recreation sites exist within the four SNC burned areas. However many opportunities for dispersed, primitive, and unconfined forms of recreation were impacted by the fires. The primary forms of recreation in the burned areas include quail hunting, pine nut collecting, heritage tourism, geocaching, hiking, and OHV

touring. Some desert bighorn sheep, mule deer, and small mammal hunting bring smaller numbers of people to these areas. There was also some temporary impact to supplemental values such as desert tortoise habitat and sensitive cultural resources, and loss of wildlife habitat likely had a negative impact on hunting and wildlife viewing opportunities. Several hunting outfitters and guides also operate in areas impacted by the fires, and a special recreation permit issued to the Second Nature Wilderness Therapy Group included areas burned in the SNC.

One ES specification was proposed by the BAER recreation team to issue a temporary administrative closure of minor OHV routes within critical and non-critical desert tortoise habitat for two years, or until stabilization treatments become established. OHV use occurs on a network of existing undesignated routes, tracks, trails and washes in addition to some cross-country travel. Loss of vegetation was expected to increase illegal cross-country travel.

SUMMARY OF BAER TREATMENT RECOMMENDATIONS

Based on aerial and ground surveys BAER team resource specialists identified the following ES treatments for implementation. These treatments are in accordance with National Emergency Stabilization and Rehabilitation (ESR) Policy and the Interagency Burned Area Emergency Stabilization Handbook.

1) Model Flow and Sediment Delivery to Beaver Dam Wash and Virgin River to determine if threats exist to downstream values at risk as a result of the Duzak Fire and develop treatments if necessary;
2) Model Flow and Sediment Delivery to Meadow Valley Wash to determine if threats exist to downstream values at risk as a result of the Duzak and Meadow Valley fires and develop treatments if necessary;
3) Replace and/or install public safety signs, on all four fires, on roads entering burned areas and where streams cross roads downstream of burned areas to inform the public of immediate danger posed by flash floods generated by storm events;
4) Aerially seed 12,857 acres on the Delamar Fire, 24,343 acres on the Duzak Fire, 7,463 acres on the Meadow Valley Fire, and 2,337 acres on the Halfway Fire in pinyon-juniper woodland and mesic blackbrush communities using primarily native perennial grasses and forbs specific to vegetation community and ecological region, in order to stabilize soils and control the spread of invasive non-native species;
5) Hand seed and rake approximately 4 miles of tracks/trails in the Delamar Fire, 15 miles in the Meadow Valley Fire, and 5 miles in the Halfway Fire with native grasses to camouflage the trails and decrease road entry, and to protect the sites from invasive non-native grass species and noxious weeds;

6) Conduct a wild horse and burro census and gather on 250 wild horses on 5 Herd Management Areas for the protection of life and property, the protection of ES treatments, and to assist in the recovery of T&E habitat and mitigate invasive plant invasion;
7) Reconstruct or extend approximately 28.5 miles of fence to maintain burn area closures and protect seeded areas and critical threatened desert tortoise habitat (ACEC) from cattle and horses;
8) Construct a 7-mile temporary exclosure fence in the Barclay Grazing Allotment within the Duzak Fire to protect critical seeded areas from burros, cattle and horses, and OHV usage;
9) Lop and scatter 200 acres on the Duzak Fire to reduce erosion effects in the watershed above Beaver Dam Wash and Meadow Valley Wash in pinyon-juniper woodlands to trap suspended sediments, control overland flow of ash and soil, and trap moisture to provide a micro-site for increased success of natural revegetation;
10) Monitor ES treatment effectiveness for vegetation recovery, seeding success, and noxious and invasive non-native weed expansion;
11) Aerially seed two essential habitat components for desert tortoise with native shrubs (for cover) and forbs (for forage) on 4,945 acres of burned desert tortoise critical habitat;
12) Apply herbicide along 113 miles of road (3,270 acres) within the SNC to inhibit germination of red brome and cheatgrass;
13) Complete known cultural resource assessments at 34 sites to assess damage/loss and post-fire risks;
14) Conduct law enforcement patrols of sites and close areas/roads until collector interest drops off and vegetation becomes re-established;
15) Consult with federally recognized tribes that have cultural ties to the area;
16) Conduct field surveys of burned areas to identify public safety hazards;
17) Provide for an Implementation Leader to coordinate all aspects of implementing, tracking and reporting on ES actions approved in the SNC ES Plan; and
18) Issue a temporary administrative closure of minor OHV routes within critical and non-critical desert tortoise habitat for two years or until stabilization treatments become established.

Two Rehabilitation (R) specifications were also identified by the BAER team:

19) Repair two damaged desert bighorn sheep guzzlers; and
20) Seed additional burned desert tortoise habitat areas identified by the Ely BLM staff.

Not all of the treatments recommended by the BAER team were approved or funded. Treatment implementation is related in greater detail in the ensuing chapters.

The eleven SNC fires burned nearly 740,000 acres in the Mojave Desert during the summer of 2005, more than the nearly 722,000 acres of fire this desert experienced in the preceding 25 years combined (Brooks and Matchett 2006). Desert ecosystems do not commonly experience wildfires and are not well adapted to recover from large fire events. The SNC was a significant ecological event for the Mojave Desert and presents many land management challenges in southern Nevada and for the Ely District BLM. The following chapters give further details on minor and major treatment implementation on the four SNC fires in the Ely BLM district, including findings on establishment of seeding treatments, post-fire vegetation trends and soil erosion risks, remote sensing assessments and applications, and lessons learned for landscape-scale fire rehabilitation in the Mojave.

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Appendix 1-1. A Statement from the Ely District BLM Fire Management Specialist

On June 22, 2005 a lightning storm moved through the Southern part of the Ely District igniting numerous fires in the Mojave Desert, the Mormon Mountains, and Meadow Valley. The Ely District BLM had anticipated high fire activity in the desert and had stationed fire crews in a temporary station (camp trailer) in the area. Ely District crews observed the lightning storm and took immediate action to suppress the resulting fires. From the start, however, the number and extreme activity of the fires overwhelmed these resources.

This extreme fire activity was a direct result of a heavier than usual fine fuel crop, due to higher than normal precipitation in the winter of 2004/2005 and spring of 2005. By the summer of 2005, the bromes in the southern Ely District were up to 2 feet tall and growing in places we had never recorded them before. The area was like a carpet of brome, extending north from Las Vegas to Caliente and east into Utah.

The Ely District Fire Management team ordered a Type II Incident Management team (IMT). They prioritized existing district resources until the Type II team arrived and was able to take over the suppression efforts of the fires. I was one of the initial Incident Commanders (ICs) managing the Meadow Valley fire.

During my time on the fire, I recorded it moving faster and with a larger front than I had ever experienced in my career, which at the time was 18 years. As the IC of the Meadow Valley fire, I decided that the only safe way we could make an effort to stop the fire was to remove the fuel ahead of it. Therefore, we attempted burn out operations in selected areas to control the fire, each time having the fire blow past our efforts.

Once the IMT arrived, they assumed suppression efforts for all fires which from there on out were referred to as the South Desert Complex (and subsequently the Southern Nevada Complex). In retrospect, I think that given the complexity and the scope of what was happening, this team was overwhelmed from the start, and; maybe the district should have ordered one or two more Incident Management teams right away, to take over the Delamar fire. This was eventually done on July 3.

As we all know it is easy to make decisions if you know what is going to happen; hindsight is 20/20!

What follows is a record of the rehab effort that was undertaken on the 700,000 + acres of the South Desert Complex including the Delamar fire.

Bill Panagopoulos
Fire Management Specialist

Chapter 2: Minor ES Treatments and BAR Treatments

INTRODUCTION

The Southern Nevada Complex (SNC) Emergency Stabilization (ES) and Burned Area Rehabilitation (BAR) plans included 27 individual specifications/treatments for 597,096 burned acres within the Ely Bureau of Land Management (BLM) District Office. A majority of the implemented ES treatments are considered minor for the purposes of reporting and analysis, and are presented here with BAR treatments in this chapter. Major ES treatments are presented in Chapters 3-9.

MINOR ES TREATMENTS

Native American Consultation (S1, 13-C)

Planned Treatment. The ES plan proposed a consultation meeting with tribal representatives from the Moapa Band of Paiutes and Paiute Indian Tribe of Utah to discuss BAER treatment plans and site concerns for the SNC fires. This treatment was proposed in accordance with federal legislation requiring that the lead agency consult with affected tribes as equal partners in federal undertakings that may affect historic properties or sites of religious significance. The plan allowed for one follow-up field consultation to discuss issues raised in the initial consultation.

Implemented Treatment. Consultations with tribal officials from the Moapa Band of Paiutes and the Paiute Indian Tribe of Utah, Band of Shivwits took place at the BAER coordination meeting in Mesquite, Nevada, in 2005. Concerns were considered during the treatment planning process.

Implementation Leaders (S1, 22-A)

Planned Treatment. The ES plan proposed funding to support an Implementation Leader, an administrative person, and an Implementation Coordinator to oversee implementation of SNC Emergency Stabilization and Rehabilitation (ESR) treatments for one year. The intent of this provision was to provide fiscal support for proper administration of the ES treatments prescribed within the SNC ES Plan, including: contract and budget administration; coordination of all ES treatments to ensure stabilization actions were achieved in a timely manner; treatment documentation for all treatments; maintaining financial cost tracking; coordinating on-the-ground implementation of treatments; supervising implementation activities; monitoring treatments for compliance with federal laws and regulations; compiling semi-annual accomplishment reports; and compiling a final accomplishment report for distribution within the United States Government and publicly on government-administered websites.

Implemented Treatment. The Washington Office (WO) decreased the total requested funding for this treatment. The WO saw no need for an Implementation Coordinator in Ely and made the determination that responsibilities could be shared between the ESR Coordinator in Las Vegas and the Team Lead in Ely.

The Ely BLM assigned duties and staff in accordance with reduced funding. The Noxious and Invasive Species Coordinator was given a temporary promotion to the Team Lead for the ESR Program. In fall 2005, a Management and Program Analyst was detailed to the ESR program to support contracting and budget administration, and in March 2006 an Administrative Support Assistant was hired to assist with the financial tracking of the SNC fires.

The Ely ESR staff oversaw all aerial seeding treatment applications for the SNC fires for both Ely and Las Vegas. Ely ESR staff was also responsible for a majority of treatment data entry for both the Ely and Las Vegas SNC fires for the first post-fire year. This included writing and submitting end-of-year accomplishment reports to the State and Washington offices.

Treatment Effectiveness. The 2005 fire season was very busy with five other fires to stabilize in the Ely District in addition to the SNC fires. Staff workloads were overwhelming, exacerbating implementation challenges. Difficulties tracking implementation and budget were common. The ESR Team Lead had to juggle two programs in addition to other special projects because it took over a year to hire a new Noxious and Invasive Species Coordinator.

The 2006 fire season was extremely busy as well. The BLM ESR team wrote and implemented 46 ESR plans in 2006, making it even more challenging to continue implementing, tracking and reporting for the SNC fires.

It took until 2008 (and beyond) to get all of the implantation tracking and reporting straightened out, and this was only possible because of slow fire seasons post 2006. Funding a dedicated Implementation Coordinator position and filling it in a timely manner could have improved the success of this treatment.

Wilderness Access Hand Seeding (S2, 8-V)

Planned Treatment. The ES plan proposed hand seeding and breaking up compacted soils on all-terrain vehicle (ATV) trails and illegal two-tracks. The purpose of this treatment was to establish vegetation in burned wilderness areas in order to stabilize soils and camouflage previous use to prevent future use.

Hand seeding was planned to conform to the wilderness area minimum tool provision and to prevent introduction of non-native invasive species into wilderness areas via motorized equipment. Due to high vegetation mortality in these areas, live seed banks were limited so that reestablishment was not likely to meet the stabilization expectations without treatment.

Decommissioning approximately 24 miles of trails and two-tracks was proposed, including 5 miles of tracks in the Halfway Fire, 15 miles of tracks in the Meadow Valley Fire, and 4 miles of tracks in the Delamar Fire.

Implemented Treatment. The wilderness access hand-seeding treatment was not approved for ES funding. This treatment was subsequently proposed in the BAR plan and approved. See treatment R2 for further information.

Post-Emergent Herbicide Application Along Roads in Burned Area (S5, 2-W)

Planned Treatment. The ES plan proposed application of a post-emergent herbicide (Roundup UltraMAX®) to 3,270 acres on the SNC to protect recovery of desert tortoise habitat from invasive non-native grass establishment—e.g. red brome (*Bromus rubens*) and cheatgrass (*Bromus tectorum*). This treatment would also provide fuel breaks to minimize probability of subsequent large-scale wildfires. Proposed herbicide application rates were 10-13 ounces per acre along a 150-foot swath on both sides of 113 miles of identified roads in the Delamar, Duzak, Halfway, and Meadow Valley fires.

The plan proposed two post-emergence herbicide applications. The first was to occur after fall precipitation events (October-November) and the second in the spring (March-April). Applications were timed to occur when target plants were small or flowering, before seeds matured and dispersed.

Implemented Treatment. This treatment was not approved for ES funding. A Noxious Weed and Invasive Plant Control and Revegetation treatment was subsequently proposed in the BAR plan and approved. See treatment R5 for further information.

Lop and Scatter (S6, 9-V)

Planned Treatment. The ES plan proposed lop and scatter on 200 acres of pinyon-juniper wood- and mixed shrub-lands within the Duzak Fire. The primary purpose of this treatment was to reduce erosion in the watershed above Beaver Dam and Meadow Valley washes by trapping suspended sediments and moisture, controlling overland flow of ash and soil, and providing microsites to increase success of natural revegetation.

Implemented Treatment. The Ely BLM lop and scatter treatment was never implemented on the Duzak Fire. The treatment was not funded, as it was not considered cost-effective given steep terrain, long drive times to treatment sites, and difficult access necessitating long carry distances for materials and equipment.

Model Flow and Sediment Delivery to Meadow Valley Wash and Beaver Dam Wash (S6, 14-WS and 17-WS)

Planned Treatment. The ES plan proposed modeling of hydrologic flows and sediment delivery from tributaries within the SNC fires to the Meadow Valley and Beaver Dam washes, including a downstream portion of the Virgin River. The purpose of this treatment was to determine if there were threats to downstream "values at risk" as a result of the Duzak and Meadow Valley fires and, if so, to develop protective treatments. "Values at risk" are any infrastructure, town, or important habitat feature that is at increased risk due to fire. Values at risk for these two fires included Union Pacific Railroad assets, roads, culverts, power lines, Threatened and Endangered (T&E) species, and BLM Sensitive species.

Implemented Treatment. Post-fire geomorphic modeling of the Meadow Valley Wash and Beaver Dam Wash watersheds was approved and contracted out to PBS&J, an environmental consulting firm in Missoula, Montana. A team of PBS&J specialists including a watershed scientist, a hydrologist, a geomorphologist, and a plant ecologist assessed hillslope processes present in both burned and unburned areas throughout the study area. They conducted three field visits, in November and December 2006 and February 2007, in which they surveyed watersheds in Cottonwood Wash, Pennsylvania Wash, the Lyman Crossing area, and the Kane Springs Road area. During these visits, PBS&J observed burn severity, soil composition, vegetation type, vegetative cover, hillslope steepness and aspect, and evidence of past erosion and severe rain events.

These field observations provided clues to the major causal factors in post-fire changes to flood hazard in the Meadow Valley and Beaver Dam washes. In addition, substantial time was spent in the Echo Fire burned area north of Panaca. This fire occurred in 2006, and the erosion dynamics here were thought to be indicative of the runoff and erosion processes which may occur immediately after wildfires in the Meadow Valley and Beaver Dam washes.

PBS&J also inventoried Meadow Valley Wash and Beaver Dam Wash watersheds for values at risk. Six smaller watersheds within these areas were chosen for Geographic Information System (GIS) modeling to prioritize revegetation areas because flooding and sediment transport events here posed the greatest threat to identified values at risk. PBS&J selected these six watersheds based on five factors: values at risk, watershed size, location of watershed, percent of watershed burned, and percent highest priority areas. Steepness and

proximity to stream were identified as two of the main drivers increasing risk of flood hazard that could be efficiently modeled in GIS. (Annual precipitation was not used as a factor in the model, as short intense summer convection storms are the driving processes in this landscape, not annual precipitation.)

Treatment Effectiveness. PBS&J provided recommendations for priority revegetation areas in the six modeled watersheds and, based on input from Ely ESR staff, proposed three seed mix alternatives for the northern portion and three seed mix alternatives for the southern portion of the SNC. PBS&J also recommended planting buffers to help reduce flooding potential.

PBS&J concluded that there were no reliable ways of accurately predicting the response of an ecosystem to a disturbance that, to the best of their knowledge, had never occurred before. In a majority of the study area, pre-fire fuel continuity was historically low, and knowledge of post-fire hydrologic impacts in the Mojave Desert and Great Basin is extremely limited. The effects of wildfires in the project area could therefore not be predicted with any degree of certainty.

A full report of findings was submitted to the BLM in September 2007. The report can be found in Appendix D.

Stateline Boundary Exclusion Fencing (S7, 6-V)

Planned Treatment. The ES plan proposed reconstruction or extension of 22 miles of the Stateline Fence impacted by the Duzak Fire on the Nevada/Utah border to protect burned area seedings from cattle and horses entering from Utah. The plan also proposed repair and reconstruction of approximately 6.5 miles of the southern Snow Springs Boundary Exclusion Fence on the Duzak Fire to prevent cattle and horses from grazing in a Desert Tortoise Area of Critical Environmental Concern (ACEC). The ACEC would normally have been protected from grazing by the existing boundary fence.

Implemented Treatment. Four fences on the Duzak Fire were either built or repaired. The Stateline Fence project was contracted out and required that three existing Stateline boundary fences totaling 8.7 miles between Nevada and Utah be rebuilt. The contract was initiated on September 20, 2006, and construction was completed on October 17, 2006. (Figure 2-1)

The southern Snow Springs Boundary Exclusion Fence was also contracted out and was completed on February 28, 2006. Approximately 6 miles of existing fence were repaired. This fence serves as an allotment boundary between the Snow Springs and Sand Hollow

allotments, as well as part of the northern boundary fence for the Beaver Dam Slope ACEC. (Figure 2-1).

Both the Stateline and Snow Springs fences are considered permanent livestock control structures.

Two other fences were reconstructed or extended under the Stateline Boundary Exclusion Fence treatment. These fences included three separate sections of the Gourd Spring Fence (Halfway and Duzak fires) and the Stratton Cross Fence (Duzak Fire). These fences were not originally proposed; however, subsequent on-the-ground reconnaissance revealed that they were a high priority for managing livestock within grazing allotments and for allowing reestablishment of vegetation.

Accordingly, the repair of approximately 4 miles of existing allotment boundary fence, between the Gourd Spring and Summit Springs allotments, was contracted out and completed on the Duzak Fire on February 28, 2006. This fence also serves as the northern boundary fence for the Beaver Dam Slope ACEC. (Figure 2-1).

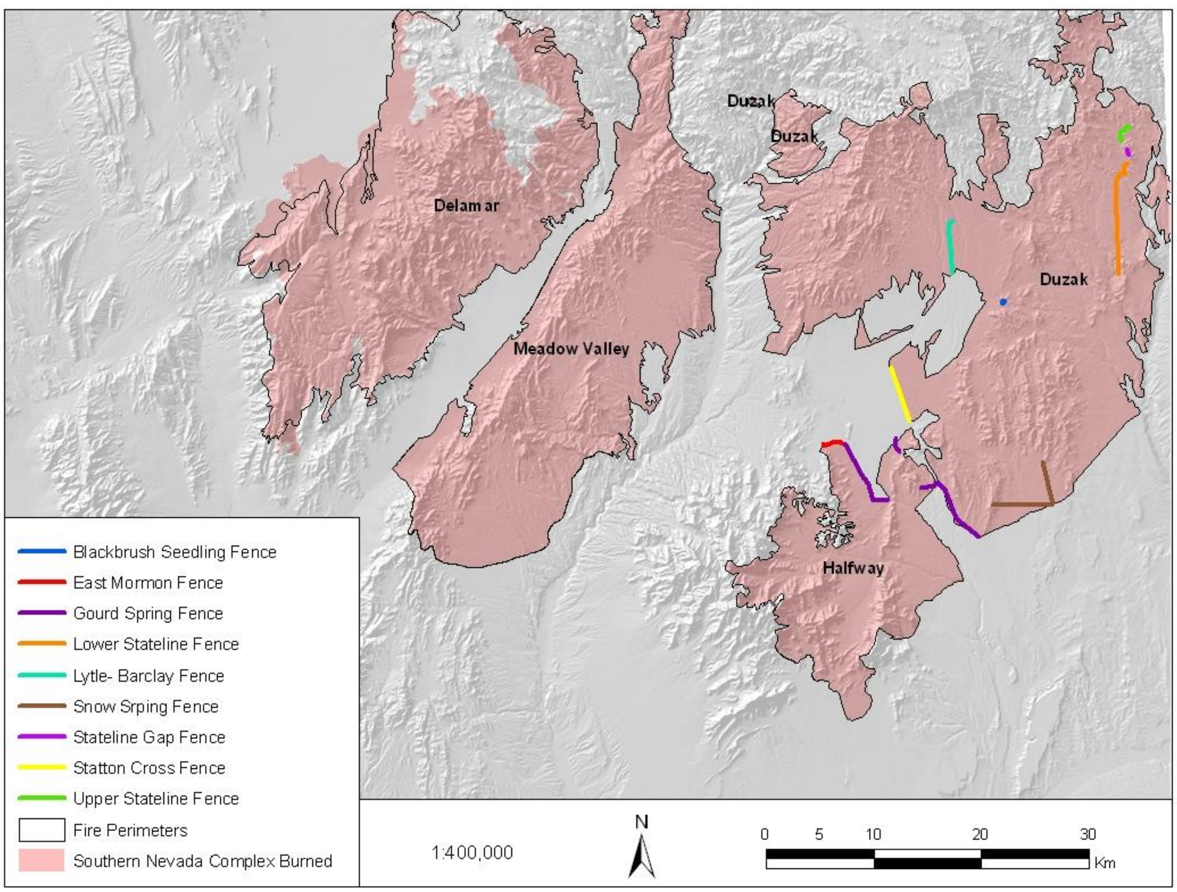

Figure 2-1. Location of fences within the Southern Nevada Complex fires.

In addition, approximately 4.5 miles of the Gourd Spring Fence was built on the Halfway Fire between March 1, 2006 and March 15, 2006. This project was a cooperative effort between the BLM and the Gourd Spring Allotment permittee. The BLM supplied the material, and the permittee supplied the labor. This temporary fence is located on the Carp/Elgin Road. The original plan was to complete this section of the Gourd Spring Fence with BAR funding. It was completed instead with ES funding because it was considered high priority for keeping cattle out of the burned area. Management has made the decision to leave the fence in place for now.

The rebuilding of approximately 2 miles of existing boundary fence between the Gourd Spring and Summit Springs allotments within the Halfway Fire was contracted out and completed on February 24, 2007.

The Stratton Cross Fence was constructed under the same contract as the 2 miles of existing boundary fence completed on the Halfway Fire. This was a newly constructed fence on the Duzak Fire in the Garden Spring Allotment. The fence is approximately 3.3 miles in length and is presently considered a permanent pasture fence.

Treatment Effectiveness. The fences built on the Halfway and Duzak fires are in good condition. Unfortunately, less than half of the needed repairs have been completed. Cattle from Utah may therefore be able to enter the burned area, and cattle from both Nevada and Utah may be able to enter desert tortoise habitat. Personnel on the burned area observed signs of livestock use including dung piles, tracks, and actual animals between 2006 and 2008 throughout the southern portion of the Duzak Fire and within mesic blackbrush (*Coleogyne ramosissima*) seeding areas.

Exclosure Fencing (S7, 7-V)

Planned Treatment. The ES plan proposed constructing 7 miles of temporary protective fence in the Barclay Allotment within the Duzak Fire burned area to protect reseeded areas from cattle and OHVs.

Implemented Treatment. This treatment was not approved for ES funding and was therefore not implemented.

Known Cultural Site Assessment (S10, 11-C)

Planned Treatment. The ES plan proposed an assessment of known National Register (NR) or potentially eligible archaeological sites in the Ely District for post-fire damage and potential risk from erosion, looting, or vandalism. The purpose of this treatment was to protect significant archaeological sites from further loss of integrity as a result of post-

fire effects and to provide recommendations for ES on easily accessible sites that were deemed highly susceptible to looting.

Known habitation sites, rock shelters and caves, rock art, and historic sites within the burn perimeter on the Ely District were considered for post-fire stabilization. As of August 2005, 34 cultural resource sites remained in need of assessment: 19 sites on the Halfway Fire, six sites on the Meadow Valley Fire, seven sites on the Delamar Fire, and two sites on the Duzak Fire.

Implemented Treatment. Post-fire damage assessment of the remaining 34 cultural sites was conducted by the Great Basin Institute (GBI). GBI completed assessments and inventories at each of the sites during March and April 2005 (Table 2-1)

Table 2-1. Known cultural sites in the SNC fires assessed for post-fire damage. Two sites within the Meadow Valley Fire and one site within the Halfway Fire were not physically inspected. These sites were inaccessible and were visually inspected with binoculars.

Site Classification	Delamar	Duzak	Halfway	Meadow Valley
Rock Art	3	0	2	4
Rock Shelters	2	2	16	0
Historic Building/Structure	2	0	1	2
TOTAL	**7**	**2**	**19**	**6**

Treatment Effectiveness. The GBI Archeological Technician and the Ely District BLM archeologists determined that none of the 34 assessed cultural sites required ES. The official report from GBI was never completed. A draft report (Final Ely District Known Archaeological Sites for the SNC) was compiled and includes all of the assessments, methods, and background information. It can be found in Appendix D.

The original draft submitted to the BLM included the locations of cultural sites. These locations have been blacked out to protect these cultural resources.

Increase Law Enforcement Patrol (S10, 12-C)

Planned Treatment. The ES plan proposed law enforcement patrols of selected cultural resource sites to enforce area closures, protect sensitive cultural resources, and detect looters. Reduced ground cover from fire-exposed cultural resources and long-abandoned roads, and special attention was recommended for resources that were subject to active trespassing, looting, or vandalism. Patrols were planned until public interest decreased and vegetative regrowth obscured previously exposed artifacts and features.

Implemented Treatment. Funding was approved and given to the BLM Las Vegas Law Enforcement Officer; however, only limited patrol of the burned areas occurred during 2006 due to workloads. Law enforcement staff was dispatched to other locales and thus

unable to spend the requested time in the SNC burned area. Other BLM staff assisted with patrols, in conjunction with their regular duties.

Treatment Effectiveness. Although additional law enforcement patrol of the known cultural sites within the burned area in the Ely District was needed, no vandalism or looting has been reported.

Wild Horse and Burro Gather (S12, 4-V)

Planned Treatment. The SNC fires burned in and around six Herd Management Areas (HMAs) within the Ely District: Meadow Valley Mountains (Meadow Valley Fire), Blue Nose Peak (Duzak Fire), Clover Creek (north of the SNC), Clover Mountains (Duzak Fire, Meadow Valley Fire), Applewhite (north of the SNC), and Delamar Mountains (Delamar Fire, Meadow Valley Fire). Pre-fire flight census data showed that these areas contained 250 wild horses. The ES plan proposed gathering all 250 horses from the six HMAs to protect ES treatments, assist recovery of critical desert tortoise habitat, and mitigate weed invasion.

Gathered horses would be processed through the BLM wild horse and burro adoption center in fiscal year (FY) 2006. In addition, post-gather census flights in the Ely District's six HMAs were proposed for FY 2006 to determine the gathers' effectiveness.

Implemented Treatment. A wild horse gather was conducted in February 2006 in the burned HMAs in the Ely District. Table 2-2 presents the number of horses gathered.

Table 2-2. Wild horses gathered from the SNC fires in February 2006.

HMA	FIRE	# OF HORSES GATHERED
Delamar Mountains	Delamar Fire	46
Meadow Valley Mountains	Meadow Valley Fire	7
Clover Mountains	Duzak Fire	11
TOTALS		**64**

Additionally, wild horses were gathered in December 2006 for the Clover Fire (2006) within the Clover Mountains HMA. Nineteen wild horses were removed. These numbers are not reflected in the table above, but are expected to benefit the Duzak Fire.

Funding for the post-gather census flights on the SNC was not approved. However, the ESR program was able to use routine flight census data collected every 1-3 years by the Wild Horse Program on all HMAs within the District (Table 2-3). This flight census data reflects the entire HMA and not just the burn area. The Wild Horse Specialist was also able to acquire additional post-gather census data in June 2007 for the northern portions of the

Duzak and Meadow Valley fires (on the Clover Mountains HMA) while conducting a flight census for the Clover Fire.

Table 2-3. Post-gather flight census summary for HMAs affected by the SNC fires.

FIRE	BURNED HMA	FLIGHT CENSUS DATE	HORSES OBSERVED DURING FLIGHT CENSUS
Delamar	Delamar Mountains HMA	Jan-07	45
Duzak	Clover Mountains HMA	Jun-07	75
Meadow Valley	Delamar Mountains HMA	Jan-07	45
	Clover Mountains HMA	Jun-07	75

Note: There was no post-gather census information for the Blue Nose Peak HMA (Duzak Fire) or the Meadow Valley Mountains HMA (Meadow Valley Fire).

In April 2007, the Nevada Division of Wildlife (NDOW) recorded 33 wild horses in the Delamar Range while conducting a flight census for spring deer. These numbers are not included in Table 2-3 above.

Treatment Effectiveness. The BLM Livestock Compliance Monitor (LCM) and vegetation monitoring crews surveyed much of the burned area while traveling many of the roads within the burn and hiking to monitoring plots. During the summers of 2006 and 2007, monitoring crews reported that signs of wild horses were common in the Delamar, Meadow Valley, and Duzak fire burn areas. Fresh dung piles were common throughout the seeding treatments. In some areas, actual animals were observed in proximity of the burned area. Even though signs of wild horses were common in the burned areas and in aerial seeding treatment areas, monitoring crews reported that the animals had not overused the burned area and regenerating vegetation was not overly grazed. In October and December 2007, wild horse sightings were noted in the Delamar Fire burn area by the livestock grazing compliance monitoring technician. Both sightings were near water sources.

In 2008, the LCM and vegetation monitoring crews reported no signs of wild horses within the Duzak or Meadow Valley fire burn areas, and wild horse signs within the Delamar Fire were uncommon. Vegetation monitoring crews and the LCM confirmed wild horse signs at two locations on the Delamar Fire, including actual animals in one location. No signs of wild horse use were reported within seeding areas.

Public Safety Hazard Assessment (S14, 20-O)

Planned Treatment. The ES plan proposed that BLM resource specialists identify hazards to public safety as a result of the SNC fires. Potential hazards included but were not limited to: abandoned mines, hazardous materials, road washouts, public safety sign damage, and any other hazards that posed a potential public safety risk. If public safety hazards were identified, a plan amendment would be written and submitted.

Implemented Treatment. Resource specialists from the Ely Field Office consulted records and maps of the entire burn area and determined that there were no known public safety hazards as a result of the fire. On-the-ground reconnaissance was not performed.

Treatment Effectiveness. In 2006 vegetation monitoring crews found the remains of a plane crash within the burned area of the Delamar Fire. Crews recorded the global positioning system (GPS) location of the crash site and gave the information to the Ely Field Office Hazardous Material Program and Nellis Air Force Base (AFB). Representatives from Nellis AFB removed all sensitive material and any potentially hazardous materials (e.g. the fuel tank) from the site. In July 2008, a BLM Environmental Protection Specialist, a BLM Rangeland Management Specialist, and two representatives from Nellis AFB returned to the site to assess removing the wreckage. Because no hazardous materials were present and the cost for removal would be significant, they decided to leave the wreckage on site. No other public safety hazards have been reported.

Replace and/or Install Public Safety Signs (S14, 15-C)

Planned Treatment. The ES plan proposed replacing and installing traffic, trail, and safety signs to prevent additional resource damage and improve public safety following the SNC fires.

Implemented Treatment. Resource specialists determined that proper warning signs were already in place from spring floods in 2005. Directional signs along main roadways were replaced as needed by BLM staff.

Treatment Effectiveness. The signs remain in place and apparently prevented additional resource damage and accidents.

Livestock Closure Agreements (S15, 7-AV)

Planned Treatment. The ES plan proposed compiling agreements and grazing decisions for livestock grazing closures on allotments affected by the SNC fires to minimize disturbances of burned areas and protect reseeded areas. Seventeen permittees on 25 grazing allotments in Lincoln County were identified. Meetings with the permittees both in a group setting and one-on-one were planned to discuss grazing impacts. The plan prescribed supervising affected grazing allotments to determine if cattle were grazing during the closure period.

Implemented Treatment. Twenty-six grazing allotments were affected by the SNC fires: four on the Halfway Fire, ten on the Meadow Valley Fire, seven on the Delamar Fire, and 11 on the Duzak Fire. (Some allotments were affected by more than one fire.) Nineteen allotments were closed by signed livestock closure agreements, one allotment was

permanently closed prior to the fires, one allotment was in non-use, and four allotments were involved in ongoing litigation and, therefore, not currently used. The permittee on the remaining allotment refused to sign a closure agreement.

In fall 2007, an LCM was hired. She drove most roads within the SNC and assessed burned area recovery. She compiled reports of her findings by fire and by allotment within the SNC.

Treatment Effectiveness. The LCM and vegetation monitoring crews surveyed the burn area while traveling many of the roads within the burn and hiking to monitoring plots. Cattle were reported on the burned areas of at least 14 allotments: Barclay, Boulder Spring, Delamar, Garden Spring, Gourd Spring, Grapevine, Lime Mountain, Lower Riggs, Rainbow, Sheep Flat, Snow Springs, Summit Springs, Terry, and White Rock. Three of these allotments occur within the Halfway Fire, four within the Meadow Valley Fire, three within the Delamar Fire, and seven within the Duzak Fire. Some of these allotments may have been reopened on a limited basis by the Range Management Specialist and may not represent trespass cattle sightings.

Repeated sightings of cattle in multiple areas on the Duzak Fire were reported by vegetation monitoring crews and the LCM, with a majority of the sightings within the mesic blackbrush seeding area. On the Delamar Fire, cattle sightings were reported to the line officer.

Many fences used to control livestock were destroyed by the fires, and most could not be replaced in a timely manner. This hampered the enforcement of the livestock closure agreements and limited their effectiveness. Feral cattle known to exist in the Meadow Valley Wash area further complicated the situation.

Temporary Administrative Vehicle Route Closure (S15, 18-R)

Planned Treatment. The ES plan proposed a temporary administrative closure for minor OHV routes in National Conservation Areas and designated critical and potential desert tortoise habitat within the burned areas of the Duzak and Halfway fires. Existing undesignated vehicle routes would be signed temporarily closed to motorized travel to promote soil stabilization; protect wildlife habitat, vegetation, and sensitive cultural resources; and allow ES treatments to establish.

Information kiosk signs were planned at key access points to the burned areas explaining the purpose and need for the temporary route closures. Patrols of the burned area during the peak recreation use times were also prescribed. The ES plan also proposed development of an information brochure describing the location and purpose of temporary closures.

Implemented Treatment. This treatment did not receive ES funding. The Wilderness Program was able to implement a portion of the project using base funds. The temporary route closures were included in the Wilderness Management plans currently being written, and wilderness boundary signs have been placed at all of the key access points.

BAR TREATMENTS

Wilderness Track Seeding (R-2, R-12)

Planned Treatment. The BAR plan proposed a team of Student Conservation Association (SCA) and Nevada Conservation Corps (NCC) crews to seed native perennial species and decompact soils along unauthorized ATV trails and two-tracks in designated wilderness areas. The purpose of this treatment was to establish vegetation in burned wilderness areas to stabilize soils and camouflage previous use to prevent future use. Hand seeding was planned to conform to the wilderness area minimum tool provision and to prevent introduction of non-native invasive species into wilderness areas via motorized equipment. Due to high vegetation mortality in these areas, live seed banks were limited so that reestablishment was not likely to meet stabilization expectations without treatment.

The BAR plan proposed decommissioning approximately 24 miles of trails via seeding and hand raking. These trails included 4 miles in the Delamar Mountain Wilderness (Delamar Fire), 15 miles in the Meadow Valley Range Wilderness (Meadow Valley Fire), and 5 miles in the Mormon Mountain Wilderness (Halfway Fire).

Implemented Treatment. This treatment was originally proposed under the ES Plan but was not approved for ES funding. It was subsequently proposed and approved under the BAR plan but not funded. However, the Wilderness Program was able to implement a portion of the project using base funding and an assistance agreement with the GBI and NCC.

A total of 8.2 miles of two-track (11 routes) were decommissioned in 2006 and 2007 within the wilderness areas burned by the SNC fires, including: 6.7 miles (seven routes) on the Halfway Fire; 1.46 miles (three routes) on the Meadow Valley Fire; and 0.06 miles (one route) on the Delamar Fire. Decommissioning tracks included using hand tools to break up soil compaction and placing dead and downed vegetation into the tracks (Table 2-4).

Table 2-4. Decommissioned two-track routes within wilderness areas burned by the SNC fires.

Wilderness Area	# of Tracks (No Seed Applied)	Miles of Tracks (No Seed Applied)	# of Tracks (Seed Applied)	Miles of Tracks (Seed Applied)	Total Tracks Decommissioned	Total Miles Decommissioned
Meadow Valley Range Wilderness (Meadow Valley Fire)	2	0.56	1	0.9	3	1.46
Mormon Mountain Wilderness (Halfway Fire)	4	5.62	3	1.08	7	6.7
Delamar Mountain Wilderness (Delamar Fire)	1	0.06	-	-	1	0.06
TOTALS	**7**	**7.32**	**4**	**1.98**	**11**	**8.22**

Of the decommissioned routes 1.98 miles were also hand seeded with a mixture of native grasses, forbs, and shrubs to establish native vegetation in the tracks (see Table 2-5). One hundred pounds of Mesic Blackbrush Hand Seeding Mix was applied to one route (0.9 miles) on the Meadow Valley Fire (within the Meadow Valley Range Wilderness) on February 18, 2006. BLM staff and Sierra Club volunteers used hand seed spreaders to apply the seed. A total of 47.5 pounds of a mesic seed mix (MM2) (Table 2-6) was applied using hand seed spreaders to three routes (1.08 miles) on the Halfway Fire (within the Mormon Mountain Wilderness) in January 2008. This treatment followed observations by wilderness monitors that people had driven on three of the decommissioned routes on the Halfway Fire. These routes were again decommissioned and seeded to establish native vegetation in the tracks.

Treatment Effectiveness. In 2008 an Ely BLM Wilderness Technician was hired to monitor vegetation reestablishment in decommissioned routes. The Wilderness Technician used hoop surveys to compare reestablishing vegetation in the tracks with vegetation outside of the tracks. The 2008 monitoring data from decommissioned routes on the Halfway and Meadow Valley fires suggest that vegetation is beginning to grow in the tracks; however, vegetation outside of the tracks is denser. The Wilderness Program plans to complete the same monitoring protocol in 2010 to further monitor the effectiveness of the treatments. As of July 24, 2008, the Wilderness Technician had not monitored the routes in the Delamar Fire.

Illegal use of existing trails and two-tracks in burned wilderness areas still occurs, and wilderness rangers continue to monitor the decommissioned routes for motor vehicle trespass 2-4 times per year.

Table 2-5. Mesic Blackbrush Hand Seeding Mix applied to 0.9 miles (one route) on the Meadow Valley Fire.

Species	BULK Lbs	PLS Lbs	BULK Lbs/Acre	PLS Lbs/Acre	Seeds/lb BULK	BULK seed/sq ft	PLS seeds/sq ft
Blackbrush *Coleogyne ramosissima*	13.60	13.28	15.46	15.09	22,400	7.95	7.76
Parry Penstemon *Penstemon parryi*	2.28	1.75	2.59	1.98	610,000	36.31	27.79
Firecracker Penstemon *Penstemon eatonii*	6.85	5.59	7.78	6.35	900,000	160.73	131.26
Shadscale Saltbush *Atriplex confertifolia*	9.13	3.13	10.37	3.56	60,585	14.43	4.95
Fourwing Saltbush *Atriplex canescens*	11.32	4.52	12.86	5.14	44,203	13.05	5.22
Pale Evening Primrose *Oenothera pallida*	22.77	17.61	25.88	20.01	700,000	415.88	321.60
Green Ephedra *Ephedra viridis*	20.45	19.42	23.24	22.07	23,545	12.56	11.93
Purple Threeawn *Aristida purpurea*	2.24	1.03	2.54	1.17	250,000	14.59	6.74
Alkali Sacaton *Sporobolus airoides*	11.36	9.31	12.91	10.58	1,750,000	518.83	425.08
TOTALS	**100.00**	**75.65**	**113.64**	**85.97**		**1194.32**	**942.32**

Note: Purple Threeawn and Parry Penstemon seeds/lb were taken from the Granite Seed website. All other seeds/lb information was taken from the USDA PLANTS Database.

Table 2-6. MM2 mesic seed mix applied to 1.08 miles (three routes) on the Halfway Fire.

Species	BULK Lbs	PLS Lbs	BULK Lbs/Acre	PLS Lbs/Acre	Seeds/lb BULK	BULK seed/sq ft	PLS seeds/sq ft
Indian Ricegrass-Rimrock *Achnatherum hymenoides*	29.93	28.7	27.7	26.6	161,920	103.00	98.93
Bottlebrush Squirreltail *Elymus elymoides*	7.13	6.2	6.6	5.7	192,000	29.08	25.19
Sandberg Bluegrass *Poa secunda*	7.13	6.2	6.6	5.7	1,046,960	158.56	138.09
Sand Dropseed *Sporobolus cryptandrus*	3.33	3.1	3.1	2.8	5,600,080	395.80	363.11
MIX TOTALS	**47.50**	**44.2**	**44.0**	**40.9**		**686.44**	**625.32**

Hand Seeding in Desert Tortoise ACEC Habitat (R2, R-7)

Planned Treatment. The BAR plan proposed hand seeding burned creosote-bursage (*Larrea tridentata-Ambrosia dumosa*) communities that showed unacceptably low native vegetation regeneration, with hand-collected seed. The purpose of this treatment was to increase establishment of annual forb seedlings and promote recovery of critical desert tortoise habitat. Seeding was planned for late summer or fall to capture optimal moisture conditions. Seeded areas would be identified from remote sensing imagery and ground surveys planned for the Meadow Valley Wilderness Area in the vicinity of Sunflower Mountain. Hand seeding was planned to conform to the wilderness area minimum tool provision.

Implemented Treatment. This treatment did not receive BAR funding and was not implemented.

Intensive Rehabilitation Islands (R-4, R-8)

Planned treatment. The BAR plan proposed seeding and planting native perennial seed and seedlings in eight 4-acre islands within burned thermic blackbrush communities. The purpose of this treatment was to ensure the successful establishment of native species and eventually provide a native seed source for surrounding areas. The island strategy was considered the most feasible way to revegetate native shrubs, including blackbrush, in the Mojave Desert. The harsh climate and the potential dominance of invasive plants required concentrated efforts that were only practical for small areas.

The BAR plan called for planting 1,600 seedlings of native shrubs in each island, including *Coleogyne ramosissima* (blackbrush), *Atriplex canescens* (fourwing saltbush), *Encelia virginensis* (Virgin River brittlebush), *Grayia spinosa* (spiny hopsage), and *Ephedra nevadensis* (Nevada ephedra). Seeded species included: *Sporobolus cryptandrus* (sand dropseed), *Achnatherum hymenoides* (Indian ricegrass), *Sphaeralcea ambigua* (desert globemallow), *Achnatherum speciosum* (desert needlegrass), and fourwing saltbush. The plan included fencing to protect the islands from livestock and rodent grazing, and removal of all noxious and invasive weeds in the islands. Installing rodent-proof fences around approximately 500 Joshua tree (*Yucca brevifolia*) resprouts was also proposed.

Implemented Treatment. BAR funding was used to buy seed and grow shrub seedlings. From 2006 through 2007, shrub seedlings were grown by the US Forest Service Lucky Peak Nursery in Idaho. In October 2006, Eastern Nevada Landscape Coalition (ENLC) staff located possible areas for islands, and in December 2006 a detailed implementation plan was written.

Unfortunately, this treatment was not prioritized for limited BAR funding in subsequent years, and the original plan was amended to accommodate limited funding received in March 2008. Rather than planting at multiple locations, efforts focused on one island in the Duzak Fire.

The BLM conducted a cultural clearance on the site in early March 2008, and seedlings were received from Lucky Peak Nursery by late March. BLM and ENLC staff began planting seedlings the last week of March and continued until the last week of April 2008.

Twelve 50 x 50 ft^2 plots were set up within the island site, with each plot containing a different treatment combination to determine which strategies work best for establishing shrub seedlings in an arid environment. Each plot was planted with approximately 154 seedlings: 70 blackbrush, 70 spiny hopsage, 10 fourwing saltbush, and 3-4 Nevada ephedra and/or littleleaf sumac (*Rhus microphylla*). Seedlings were planted 2-3 feet apart.

Six plots were assigned to rock mulch treatments, and six plots were assigned to tree shelter treatments to protect seedlings from herbivory. BLM staff also constructed a 0.5-mile temporary fence around all seedlings to further protect them from herbivory (Figure 2-1).

Plots were further assigned to one of four treatments intended to increase seedling-available moisture: deep pipes, divots, deep pipes and divots together, or no enhancement. Rock mulch and tree shelters were also anticipated to increase seedling-available moisture. The BLM Recreation Technician watered the seedlings multiple times after planting.

Treatment Effectiveness. Initially, the fourwing saltbush and spiny hopsage seedlings did extremely well. The BLM Recreation Technician later reported that grasshoppers subsequently ate all of the leaves of the spiny hopsage plants.

Blackbrush seedling survival was low overall, although 32 seedlings survived and continue to grow. Many of the blackbrush seedlings were moldy when they arrived from Lucky Peak Nursery. Regardless, the moldy seedlings were planted, but in most cases they died. The seedlings that survived were healthy when they arrived, suggesting that blackbrush seedling survival can be improved if transport and storage methods are improved.

Of all treatments, the tree shelters were the most successful aid to seedling establishment. The BLM Recreation Technician continued to water the seedlings at least once each month as of June 2008. Tree shelters were removed in June 2010.

Wildlife Water Source Rehab (R-4, R-11)

Planned Treatment. The BAR plan proposed planting native perennial shrubs at Cave Spring as well as at seven small-volume water developments to reestablish critical wildlife habitat and cover. Under the proposed plan, the BLM would provide the seedlings and NDOW would implement and monitor the treatment.

Implemented Treatment. The proposed treatment did not receive BAR funding. Instead, NDOW funded and implemented all portions of this treatment. The NDOW water development crew spearheaded shrub planting with assistance from other NDOW biologists and a forester from the Nevada Division of Forestry. On November 3 and 16, 2006, shrubs were planted at two guzzlers in the Halfway Fire burn area. Shrubs were planted at three guzzlers in the Meadow Valley burn area on May 3, 10, and 17, 2007.

Treatment Effectiveness. In FY 2008, NDOW revisited all of the sites for photo documentation and to install DriWater® gel packs to provide continuous water to the shrub seedlings. Planted shrubs survived and continue to grow.

Noxious Weed and Invasive Plant Control and Revegetation (R5)

Planned Treatment. The BAR plan proposed funding one temporary "weed sentry" position to continually inventory the 597,096 acres of BLM lands burned in the SNC fires for non-native invasive weeds. Identified populations of noxious and invasive plant species would be spatially referenced and entered into a GIS database. Additionally, the weed sentry would plan and oversee weed removal and revegetation treatments in these areas. The plan estimated at least 300 one-acre weed removal and revegetation treatments would be needed on small isolated patches over three years. Weed removal would involve any appropriate combination of chemical, mechanical, and hand control methods and would be performed by contracted crews. Revegetation would involve planting and irrigating sites that had been cleared of weeds. Treated sites would be spatially referenced and entered into a GIS database.

Implemented Treatment. This treatment was not prioritized for limited BAR funding and was not funded. Weed duties were therefore split between the Weed Coordinator, the LCM, and vegetation monitoring crews, and some weed control has occurred in the SNC fires. Weed treatments have been funded by the Noxious and Invasive Species Program and implemented along existing roads in the district through an assistance agreement with Tri-County Weed. Some of these roads occur within the burned area.

Tri-County Weed monitored or treated 122 points on the Delamar Fire between July 2005 and July 28, 2008. The Bishop Spring drainage was the main treatment area with 29 specific treatment/monitoring points targeting three species: *Lepidium latifolium* (tall whitetop), *Tamarix ramosissima* (salt cedar), and *Centaurea stoebe* (spotted knapweed).

Seven locations of salt cedar were treated with the herbicides Arsenal and Garlon 4; 71 locations of tall whitetop were treated with the herbicide Escort; and no weeds were present at 40 locations. Musk thistle (*Carduus nutans*) was treated with the herbicide Tordon 22K at one location in 2005.

Between June 2005 and March 17, 2008, two general areas were monitored on the Duzak Fire. Two species—tall whitetop and Scotch thistle (*Onopordum acanthium*)—were targeted at six treatment locations. Three locations of Scotch thistle were treated with Tordon 22K; one location of tall whitetop was treated with Escort; and no weeds were present at two locations.

In 2008, one infestation of tall whitetop was located in the Meadow Valley Fire. The infestation was treated and bladed over immediately following treatment.

In November 2006, the previously halftime Noxious and Invasive Weed Coordinator position became a fulltime position. The Weed Coordinator is presently coordinating with the Las Vegas Field Office, the US Fish & Wildlife Service, and other agencies to examine herbicide use for annual grasses in desert tortoise habitat. Additionally, the Weed Coordinator is currently participating in regional planning efforts for annual grass management.

Treatment Effectiveness. Tri-County Weed has monitored the Delamar and Duzak Fire treatment locations multiple times. As of July 2008, noxious weed populations had expanded at only one location on the Duzak Fire. None of the treatment points on the Delamar Fire showed any post-fire weed population increases. There was one infestation of tall whitetop on the Meadow Valley Fire in 2008. The infestation was along the Meadow Valley Wash road and is most likely a result of the road rather than the fire.

On the Duzak Fire, the Scotch thistle infestation at Bracken Pond, near the Utah border, has increased in size and density since the fires—from 1 acre with 2-25% cover, to 2.5 acres with more than 50% cover. This infestation was first inventoried in 2004, and at that time the infestation was an estimated 500 feet^2. Just prior to the fire, in April 2005, the infestation had increased to 1 acre. The Noxious and Invasive Weed Coordinator speculated that the area was probably heavily grazed, increasing the infestation of Scotch thistle prior to the fire. After the infestation burned, it is probable a large portion of the seedbank was released.

Seed Collection for Desert Tortoise Areas of Critical Environmental Concern (ACECs) (R6, 8-V)

Planned Treatment. The BAR plan proposed collecting seed from plant species important for the recovery of burned, federally designated critical desert tortoise habitat. Contract crews would collect mature native seed from creosote-bursage plant communities in southern Nevada each year for three years. Collections would be located outside ACECs and designated wilderness areas, and crews would use standard BLM protocols to collect seeds from plant populations. Each year up to 50 bulk pounds of each desired species would be collected. The objective was to collect enough viable seed to plant up to 200 acres of burned ACEC each year.

Implemented Treatment. The Ely Field Office only received a small amount of BAR funding for seed collection, so this treatment was implemented by the Las Vegas Field Office. The Las Vegas Field Office awarded a contract to hand collect seed to the Native Seed Company in FY 2007. The Ely BLM and ENLC staff mixed the hand-collected seed in February 2008.

Treatment Effectiveness. In February 2008, the Ely Field Office mixed 2,419 pounds of hand-collected seed. The Ely Field Office received 80 pounds of this mix which included big galleta (*Pleuraphis rigida*), creosote bush, desert globemallow, desert marigold (*Baileya multiradiata*), fourwing saltbush, Indian ricegrass, needle grama (*Bouteloua aristidoides*), littleleaf ratany (*Krameria erecta*), Virgin River brittlebush, and bursage. Due to the limited BAR funding received for this treatment, this seed has not yet been applied. However, the Ely District maintains its intention to seed critical desert tortoise habitat with this mix.

Minor Facilities Repair and Replacement (R-7, R-2)

Planned Treatment. The plan proposed replacing or repairing 72.5 miles of fence and four corrals damaged by the SNC fires prior to allowing normal livestock grazing operations to recur in the rehabilitated areas. The purpose of this treatment was to ensure proper control and management of livestock grazing.

Implemented Treatment. BAR funding received for this treatment in FY 2006 was used for fence assessments in the Delamar and Meadow Valley fires. Due to limited BAR funding in subsequent years, fence repair or replacement on the Delamar and Meadow Valley fires was not implemented.

Tortoise activity, consultation with the US Fish & Wildlife Service, and staffing limitations delayed repairs proposed for 12 miles of burned wooden fences and corrals on the Duzak Fire. Three miles of fence repair on the Lytle-Barclay Fence within the Duzak Fire were contracted out and completed on October 26, 2007 (Figure 2-1).

On the Halfway Fire, approximately 4.5 miles of temporary fence were built using ES funding (see S7, Stateline Boundary Exclusion Fencing) (Figure 2-1).

Treatment Effectiveness. Personnel at the Duzak Fire observed signs of livestock use including dung piles, tracks, and actual animals in 2006-2008 throughout the southern portion of the burned area, including within mesic blackbrush seedings.

Wild Horse Census (R12, R-1)

Planned Treatment. The BAR plan proposed conducting wild horse censuses on the six HMAs in the Ely District in the second and third year (FY 2007 and FY 2008) to determine if horses or burros had moved into the burned area. If so, a plan amendment to gather and remove the animals would be submitted. The treatment objective was to prevent a permanent presence of horses or burros in the burned areas.

Implemented Treatment. This treatment was not prioritized for limited BAR funding; however, census flights were performed in the burned area between 2005 and 2007, as part of the Wild Horse Program’s regular census flights performed every 1-3 years on all HMAs in the District.

In January 2007, a flight census of the Delamar Mountains HMA (Delamar, Halfway and Meadow Valley fires) documented 45 horses. In a June 2007 flight census, 78 wild horses were observed in the Clover Mountains HMA (Duzak, Halfway and Meadow Valley fires), and seven wild horses were seen in the Clover Creek HMA (Duzak Fire). It should be noted that flight census data reflect the entire HMA and not just the burn area.

Wildlife Water Developments Repair (R14, R-3)

Planned Treatment. The BAR plan proposed repairing or replacing two 1,800-gallon tanks and replacing approximately 1,750 feet of 2-inch poly pipe to make the Meadow Valley #4 and Delamar #3 guzzlers functional again. The guzzlers were constructed in 2001 and 2002, respectively, to provide a water source for desert bighorn sheep and other wildlife in the Delamar and Meadow Valley mountains.

Implemented Treatment. This treatment did not receive BAR funding. Instead, funding used to complete the project came from NDOW, the Fraternity of the Desert Bighorn, and Heritage Funds. This project was organized by NDOW.

On July 27, 2005, the Fraternity of the Desert Bighorn funded a flight to assess the damage to the Delamar #3 and Meadow Valley #4 guzzlers and to determine the materials needed to repair them. On April 20, 2006, volunteers and a NDOW biologist repaired the guzzlers. Five hundred feet of poly pipe was replaced, painted, and buried for the Meadow

Valley guzzler, and the tank was replaced. On the Delamar guzzler all 800 feet of the poly pipe was replaced, and a hole in the top of the tank was repaired.

Treatment Effectiveness. This treatment was monitored by Wilderness Staff in January 2008. Repairs were effective, and the guzzlers are functioning properly.

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Chapter 3: Effectiveness of Post-Fire Seeding in Desert Tortoise Critical Habitat Following the 2005 Southern Nevada Fire Complex

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ABSTRACT

In June 2005, lightning strikes ignited multiple wildfires in southern Nevada. The Southern Nevada Fire Complex burned more than 32,000 acres of designated desert tortoise Critical Habitat and an additional 403,000 acres of Mojave Desert habitat characterized as potentially suitable for the tortoise. Mortalities of desert tortoises were observed after the fires, but altered habitat is likely to prolong and magnify the impacts of wildfire on desert tortoise populations. To accelerate the re-establishment of plants commonly used by tortoises for food and shelter, the Bureau of Land Management (BLM) distributed seeds of native annual and perennial species in burned areas within desert tortoise Critical Habitat. The U.S. Geological Survey (USGS) established monitoring plots to evaluate broadcast seeding as a means to restore habitat and tortoise activity compared with natural recovery. Within the standard three-year Emergency and Stabilization Response (ESR) monitoring timeline, seeding augmented perennial seed banks by four to six-fold within a year of seed applications compared with unseeded areas. By the end of the three-year monitoring period, seedling densities of seeded perennial species were 33% higher in seeded areas than in unseeded areas, particularly for the disturbance-adapted desert globemallow (*Sphaeralcea ambigua*) and desert marigold (*Baileya multiradiata*). Seeded annuals, in contrast, did not increase significantly in seed banks or biomass production, likely due to low seeding rates of these species. Production of non-native annuals that helped carry the fires was not reduced by seeding efforts but instead was strongly correlated with site-specific rainfall, as were native annual species. The short-term vegetation changes measured in seeded areas were not yet associated with a return of tortoise activity to unburned levels. By focusing on a combination of native species that can withstand disturbance conditions, including species that are found in adjacent unburned areas, and increasing seeding rates, broadcast seeding has strong potential to provide herbaceous plants for forage and long-term perennial plant cover to support tortoise recovery in burned habitats.

INTRODUCTION

The Mojave Desert population of the desert tortoise (*Gopherus agassizii*) was listed as a threatened species in 1990 (U.S. Fish and Wildlife Service 1990). As part of the recovery effort for this population, land managers designated regions of critical habitat with the physical and biological attributes essential for long-term survival of the desert tortoise (USFWS 1994). In spite of related management efforts, desert tortoise populations continue to face many threats (Esque *et al*. 2003, Tracy *et al*. 2004). Wildfire, once considered a rare event in the Mojave Desert (Humphrey 1974), now occurs with greater frequency, threatening tortoises and their habitat (USFWS 1994, Brooks and Esque 2002). Fire in the Mojave Desert has increased with the proliferation of introduced grasses, particularly brome grass (*Bromus madritensis* ssp. *rubens* and *B. tectorum*) and Mediterranean grass (*Schismus*

barbatus and *S. arabicus*), which provide the fine fuels needed to carry wildfire between shrubs (Brown and Minnich 1986, Brooks and Matchett 2006).

In June 2005, lightning strikes ignited multiple wildfires in southern Nevada. These fires, termed the Southern Nevada Fire Complex, burned more than 32,000 acres of designated desert tortoise Critical Habitat and an additional 403,000 acres of Mojave Desert habitat characterized as potentially suitable for the tortoise. Wildfires can result in direct injury and mortality of desert tortoises (Esque *et al*. 2003, Nussear 2004). Indirect effects such as the loss of perennial plants used for cover from predators and environmental extremes (Duck *et al*. 1997) and shifts in the dominant vegetation from native shrubs to invasive grasses (Brooks *et al*. 2007) may have a more profound and lasting impact on the survival of this species following wildfire (Figure 3-1). These changes may result in a decline in growth and reproductive output (Duck *et al*. 1997, Brooks and Esque 2002) or cause tortoises to emigrate away from burned habitat (Esque *et al*. 2003). The indirect effects of fire on desert tortoise populations, however, have not been quantitatively analyzed (Esque *et al*. 2003).

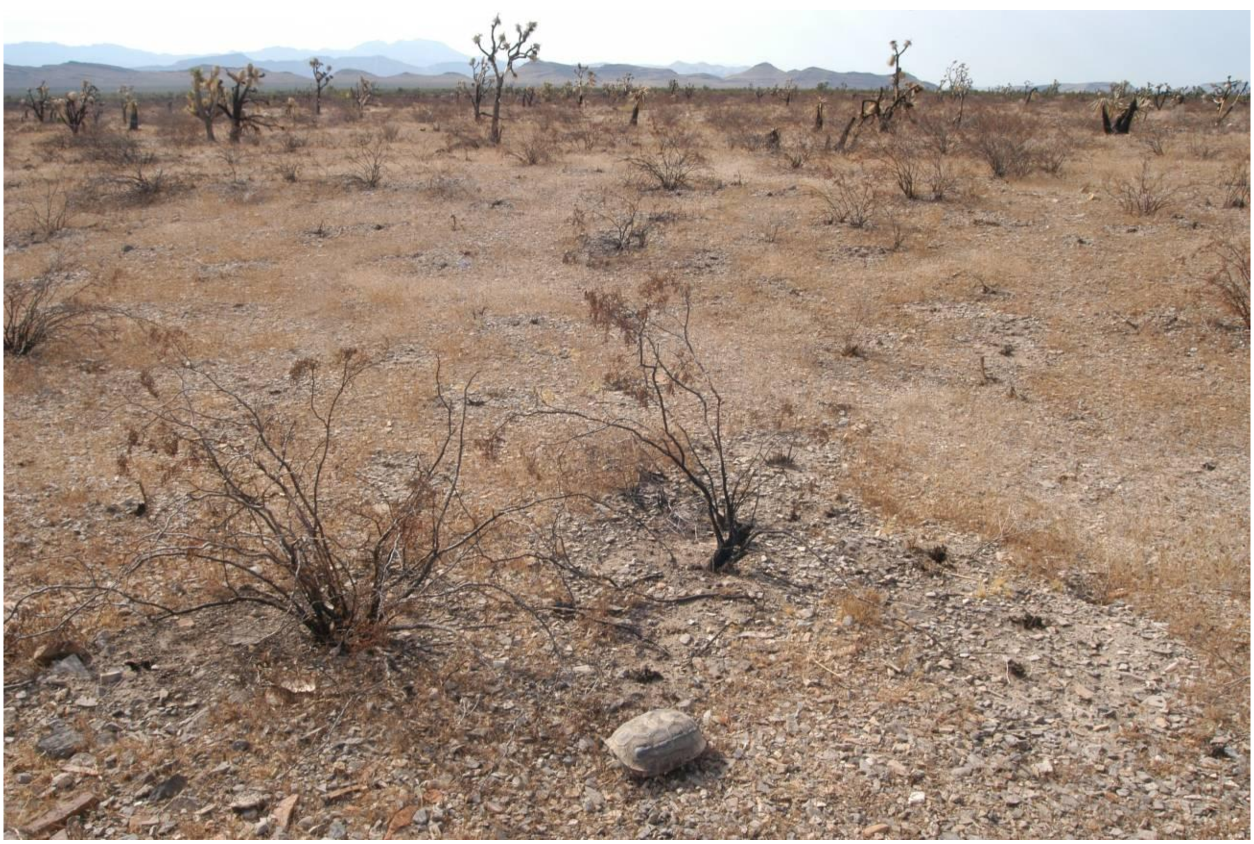

Figure 3-1. A desert tortoise is faced with a dramatically altered habitat following the 2005 Dry Rock Fire (Photo: L. A. DeFalco).

The Burned Area Emergency Response (BAER) Plan, approved shortly after the 2005 wildfires, directed the Bureau of Land Management (BLM) to seed 17,102 acres of burned desert tortoise Critical Habitat. Broadcast seeding in arid lands is challenging in part because seeds residing on the soil surface are vulnerable to loss while they wait months or, in some cases, years for optimal conditions to germinate and establish. Potential sources of seed loss include seed predators such as rodents and ants (Roth and Vander Wall 2005, DeFalco *et al.* 2009) and lateral and horizontal redistribution of seeds by wind and surface flow (Chambers *et al.* 1991, Chambers 1995). The low rate of plant establishment following past seeding efforts in the Mojave Desert is most often attributed to the region's low annual rainfall, averaging 100 mm, which provides limited opportunities for seeds to germinate and establish (Anderson and Ostler 2002, Bainbridge 2007, Caldwell *et al.* 2009).

Reviews of Mojave Desert restoration efforts offer advice on various techniques that manipulate soil moisture and contours to improve seeding success, but the suggestions are based on disturbances where military, mining, or agricultural uses have altered the structure of the soil itself (Anderson and Ostler 2002, Bainbridge 2007, Caldwell *et al.* 2009). Wildfires in the Mojave Desert do alter soil nitrogen levels and viable seed bank densities (Esque 2004), but rarely produce the level of soil compaction and topsoil loss that inhibits secondary plant succession on heavily-used land (Prose *et al.* 1987, Lovich and Bainbridge 1999, Webb 2002, Scoles-Scuilla and DeFalco 2009, DeFalco *et al.* 2009). If the physical integrity of the soil is maintained after wildfires, native seeds in the Mojave Desert are well-adapted to persist through drought conditions, with annual species employing a "bet-hedging" strategy (Brown and Venable 1986) and perennial species commonly displaying physical or physiological dormancy (Baskin and Baskin 1998). To rehabilitate burned desert tortoise habitat while minimizing further damage to plants, soil, and ground-dwelling wildlife that occurs with mechanized approaches, the practice of broadcasting native seed is likely the most viable option over large areas. Low-intensity techniques, such as pitting with hand tools, can be used to bury applied seed in hopes of enhancing seedling establishment (Bainbridge 2007), but their effectiveness in burned Mojave Desert habitat is not known, and such practices may be cost prohibitive over large areas.

The BLM (Las Vegas and Ely Field Offices) developed and distributed a native seed mix within seven fires in desert tortoise Critical Habitat in Southern Nevada composed of annual forb and grass species known to comprise > 90% of desert tortoise diet (Esque 1994, DeFalco 1995) and perennial shrub species that provide shelter from temperature extremes and predation (Burge 1978, Berry and Turner 1986). The resulting species composition of the seed mixture was based on commercial availability. At the time of seeding, the U.S. Geological Survey (USGS) established additional monitoring plots adjacent to each seeded plot to determine seeding effectiveness and compare recovery to unburned vegetation. Corresponding surveys for sign of desert tortoise activity on these plots were conducted to

examine differences in desert tortoise presence in response to the seeding treatment. We monitored vegetation and tortoise responses for three years after burned habitat was seeded to determine whether: 1) non-native annual plant production was reduced and native annuals increased on sites seeded with native species, 2) perennial plant density and canopy cover were augmented by seeding, and 3) tortoise activity, as indicated by detection of recent tortoise sign (live tortoises, active burrows, fresh scat, and tracks), increased in seeded areas. We also monitored seed banks to determine if viable seeds from the seed mix persisted one and two years following application, and we monitored monthly precipitation to evaluate vegetation responses among the broadly distributed sites in southern Nevada.

MATERIALS AND METHODS

Study Area

The study area encompassed seven fires within the Southern Nevada Complex that included designated critical habitat for the threatened desert tortoise (USFWS 1994; Figure 3-2). The Tramp and Bunkerville fires occurred in the Gold Butte-Pakoon Desert Wildlife Management Area (DWMA), while the Dry Middle, Dry Rock, Garnet, Meadow Valley, and Halfway fires occurred in the Mormon Mesa DWMA. All seven fires occurred in low elevation habitat dominated by creosote bush (*Larrea tridentata*)/bursage (*Ambrosia dumosa*)/Joshua tree (*Yucca brevifolia*) shrubland. Research plots were of similar ecological types based on soil, hydrology and vegetation (BLM Ely Field Station, pers. comm.).

Emergency-Stabilization Response

During mid-December 2005, the Las Vegas and Ely BLM Field Offices located monitoring plots within the Southern Nevada Fire Complex and seeded them with native Mojave Desert species. A total of fifteen 40-acre plots (400 m × 400 m) were positioned within the Dry Middle, Dry Rock, Garnet, Halfway, Meadow Valley, and Tramp fires and one additional 25-acre plot (250 m × 400 m) was placed in the Meadow Valley Fire (Figure 3-2). The smaller 25-acre plot at the Meadow Valley Fire was necessary due to the constraint of fitting larger plots within this patchy fire that occurred along a bajada. No seeding occurred at the Bunkerville Fire in 2005. Seed was applied to all plots by hand instead of using heavy machinery to avoid potential injury to tortoises in underground burrows. Using Pulaski and McLeod hand tools, field crew members first created shallow depressions (15 cm × 10 cm × 5 cm) at an approximate density of 100 pits/acre with the intent to capture wind-blown seed. Immediately after pitting, crew members hand-seeded each plot at a rate of 3 lbs pure live seed/acre. The seed mix was composed of four native perennial shrub and forb species (Table 3-1). Seed that became available at a later date was distributed across the same sixteen plots plus one additional 40-acre plot at the Bunkerville Fire in early November 2006. The 2006 seed mix included 8 perennial shrub, forb and grass species and 5 annual forb and grass species (Table 3-1). The seed mixture applied at the Bunkerville Fire included the same

annual and perennial species as the 2006 seed mix plus purple three-awn (*Aristida purpurea*; Table 3-1). During the 2006 seeding, field crew members hand-seeded plots at a rate of 8.5 lb pure live seed/acre (6.6 lb PLS/acre at Bunkerville). During hand-seeding, crew members followed pre-marked transects 5-10 m apart and carried pre-weighed seed bags to ensure uniform distribution of seed (Figure 3-3). Plots were not re-pitted in 2006 because pits were indistinguishable from natural surface topography within days of treatment and we did not want to damage any seedlings that had established following the first seeding.

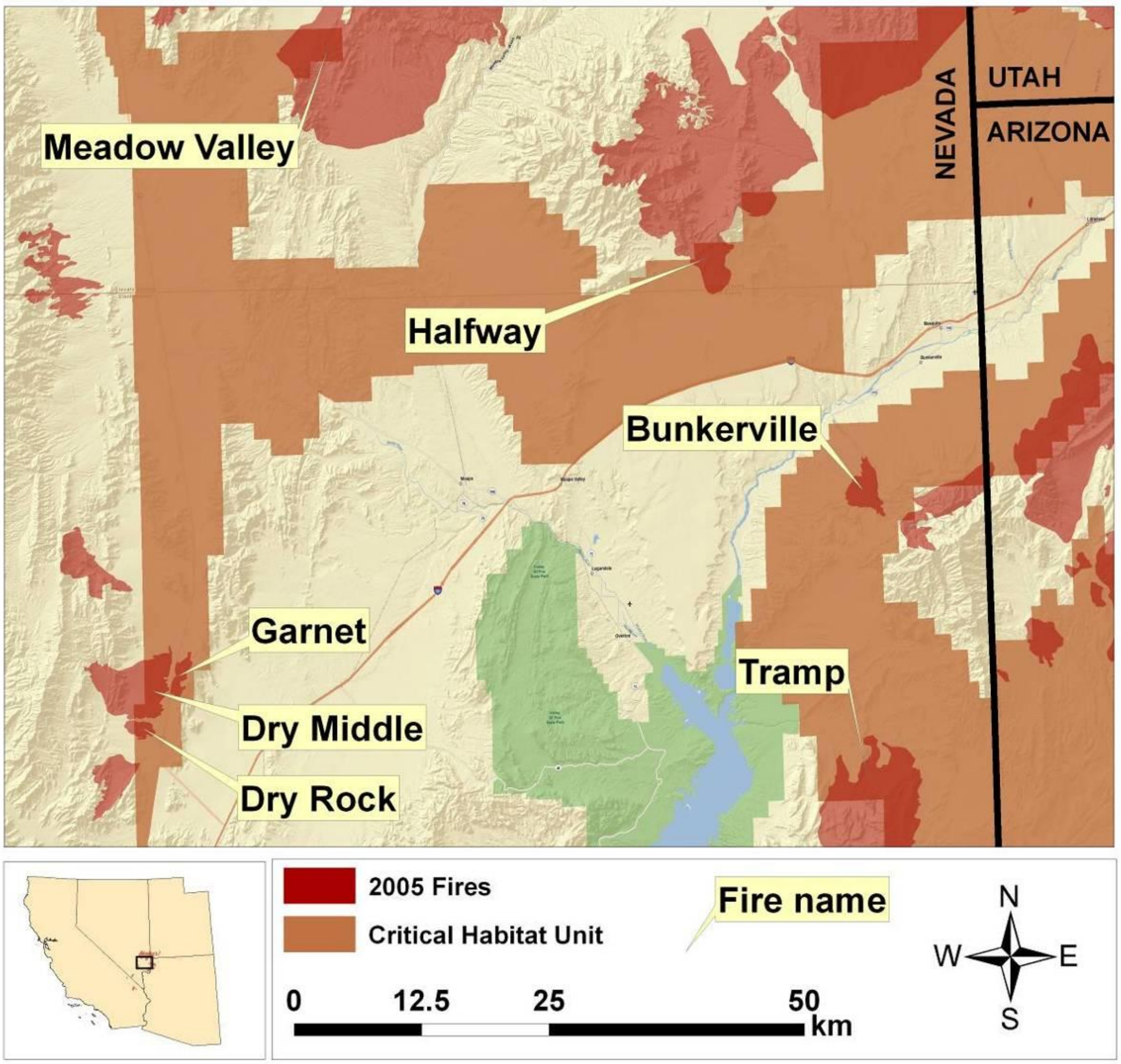

Figure 3-2. Annual and perennial vegetation and tortoise sign were monitored across seven fires that burned in designated desert tortoise Critical Habitat during the summer of 2005. BLM established burn-seeded monitoring plots in December 2005, and USGS subsequently selected burned-unseeded and unburned reference sites for monitoring in 2006–2008.

Table 3-1. Application rate of seed mix by species in 2005 and 2006. The seed mix was applied at a rate of 3 lbs pure live seed (PLS)/acre in 2005 and 8.5 lbs PLS/acre in 2006 at the Garnet, Dry Rock, Dry Middle, Meadow Valley, Halfway and Tramp fires. At the Bunkerville fire, the seed mix was applied at a rate of 6.6 lbs PLS/acre in 2006 only.

	Species	Common name	Lbs PLS/acre	Live seeds/m^2
2005	*Sphaeralcea ambigua*	Desert globemallow	1.4	175
	Baileya multiradiata	Desert marigold	0.5	122
	Ambrosia dumosa	Bursage	0.4	9
	Atriplex canescens	Four-wing saltbush	0.7	8
2006	*Bouteloua aristidoides*	Needle grama	3.7	381
	Sphaeralcea ambigua	Desert globemallow	1.5	181
	Baileya multiradiata	Desert marigold	0.6	146
	Vulpia octoflora	Six-week fescue	0.1	30
	Achnatherum hymenoides	Indian ricegrass	0.7	27
	Atriplex canescens	Four-wing saltbush	1.2	16
	Hymenoclea salsola	Cheesebush	0.4	11
	Malacothrix glabrata	Desert dandelion	0.02	10
	Camissonia claviformis	Brown-eyed primrose	0.03	4
	Larrea tridentata	Creosote bush	0.2	4
	Eschscholzia minutiflora	Pygmy goldenpoppy	0.02	2
	Ambrosia dumosa	Bursage	0.1	2
	Encelia virginensis	Brittlebush	0.02	< 1
Bunkerville	*Aristida purpurea*	Purple three-awn	2.7	166
2006	*Sphaeralcea ambigua*	Desert globemallow	1.3	156
	Baileya multiradiata	Desert marigold	0.5	127
	Vulpia octoflora	Six-week fescue	0.1	26
	Atriplex canescens	Four-wing saltbush	1.0	13
	Bouteloua aristidoides	Needle grama	0.1	10
	Achnatherum hymenoides	Indian ricegrass	0.2	9
	Hymenoclea salsola	Cheesebush	0.3	9
	Malacothrix glabrata	Desert dandelion	0.01	8
	Camissonia claviformis	Brown-eyed primrose	0.02	4
	Larrea tridentata	Creosote bush	0.2	3
	Eschscholzia minutiflora	Pygmy goldenpoppy	0.02	2
	Ambrosia dumosa	Bursage	0.1	1
	Encelia virginensis	Brittlebush	0.02	< 1

After the 2005 seeding (2006 seeding at the Bunkerville Fire), we established 40-acre monitoring plots in burned-unseeded and unburned areas adjacent to each seeded plot. These plots were selected based on similarity to burned-seeded monitoring plots (*i.e.*, soil type, vegetation community, elevation and slope) to facilitate pair-wise comparisons. Manual rain gauges (Tru-Chek® rain gauge, Edwards Manufacturing Company) were installed at the northwest corner of each monitoring plot to record monthly precipitation. Gauges were mounted on short wooden stakes and filled with 2–3 mm of clear mineral oil to reduce evaporation. A fine mesh screen was placed over the top of each gauge to prevent insects, lizards, small mammals, or debris from becoming trapped inside.

Figure 3-3. Crew members from the Nevada Conservation Corps systematically broadcast native seeds across a 40-acre monitoring plot within the Tramp Fire (Photo: L. A. DeFalco).

Changes in Annual and Perennial Vegetation

Biomass of native and non-native annual plants was measured each spring from 2006 to 2008, and biomass of seeded annual species was measured during spring in 2007 and 2008. Annual grasses and forbs were sampled during peak production (15 May to 2 June 2006, 5–19 April 2007, and 4–22 April 2008). Each year thirty 1 m^2 sampling quadrats were randomly placed within the central 300 m × 300 m area of each monitoring plot to reduce edge effects. We identified all species in each 1 m^2 quadrat and nested within it a 20 cm × 50 cm quadrat to sample annual plant shoot biomass. Shoots were clipped at ground level and sorted into native and non-native species. In 2007 and 2008, seeded annual species were also sorted. Samples were dried in a convection oven at 50 °C to a constant mass and weighed.

Perennial plant cover and density of seeded species were measured on three permanent 100 m line transects located within each monitoring plot. Transects started 10 m from the center of each monitoring plot and were oriented at 60°, 180°, and 300° from north (Herrick *et al.* 2005). Perennial plant cover was measured as canopy intercept along the transect lines, with gaps in an individual plant canopy of less than 10 cm recorded as continuous cover. Plant status was noted as dead, live, resprout, or unburned. Density of seeded species was measured within 2 m × 100 m belts centered along each transect line, with individuals classified as either adult or seedling depending on size and reproductive maturity. Perennial plant measurements were conducted once a year following annual plant sampling (26 June to 13 July 2006, 11–26 June 2007, and 19 May to 9 June 2008).

Persistence of Seeded Species in the Seed Bank

We sampled the seed bank of seeded and unseeded-burn monitoring plots for two reasons: 1) to ensure that treatment effects could be evaluated even if rainfall during the 3-year monitoring period was insufficient to promote plant establishment, and 2) to determine if viable seeds from the seed mix remained detectably higher in the seeded area compared with unseeded-burn plots. We sampled the seed bank one year following each seeding treatment (26 September to 12 October 2006 and 17–21 September 2007). Perennial grasses and shrubs can be under-represented in seed bank samples collected with standard soil cores (Esque 2004), so we used modified gas-powered leaf blowers to vacuum the surface of replicated 4 m^2 quadrats in the burn-seeded and burn-unseeded plots (n = 16 monitoring sites × 2 treatment plots/site × 10 quadrats/monitoring plot = 320 quadrats in 2006 and n = 17 monitoring sites (Bunkerville added) × 2 treatment plots/site × 10 quadrats/monitoring plot = 340 quadrats in 2007). This method has been used successfully to sample perennial and annual Mojave Desert species (DeFalco *et al.* 2009). Before vacuuming, we lightly raked the soil surface with leaf rakes to a depth of 2 cm to dislodge seeds attached to soil surfaces. Leaf blowers were modified with a mesh screen over the front nozzle to prevent large rocks and dead vegetation from entering and a 1-gallon paint strainer bag attached inside the blower bag to separate seed and fine organic debris from larger debris. In 2007 we also collected 4 soil cores (10.16 cm diameter × 2 cm deep) located outside each corner of the vacuumed quadrats to measure annual seed bank. The four soil cores were combined into one sample per quadrat (4 cores × 81.07 cm^2 = 324.29 cm^2). Seed bank samples were grown out in a greenhouse using methods modified from Young and Evans (1975) and Young *et al.* (1981). Samples were subjected to four alternating wet-dry cycles known to promote germination of seeds (Mayer and Poljakoff-Mayber 1982, Baskin and Baskin 1998), including those of Mojave Desert species (Esque 2004, DeFalco *et al.* 2009, Scoles-Sciulla and DeFalco 2009). Emerging seedlings were counted and harvested after identification using a seedling library developed for seed bank studies (USGS unpublished data).

Tortoise Sign

Full-coverage surveys were conducted each year on all monitoring plots to record evidence of tortoise activity during spring when desert tortoises were active (K. Drake, unpublished data). Survey teams consisted of 4 to 9 personnel spaced at 15-25 m intervals (Nussear *et al.* 2008) with transect passes of 400 m. The locations of all tortoise sign encountered, including live tortoises, carcasses, scats, eggshells, fresh tracks and cover sites such as burrows, caves and pallets, were recorded using GPS-enabled PDAs. Cover sites were characterized as active or inactive. All scat encountered was collected to distinguish fecal matter deposited in subsequent years. Fresh scat was recognized by a glossy and grainy surface (sometimes moist) with some fiber visible, whereas old scat had a rough, dry surface and brittle texture due to weathering.

Statistical Analyses

The effectiveness of the seeding treatment was tested by comparing vegetation and tortoise response variables from untreated and treated burned plots using SAS statistical software (version 9.1, Copyright © 2004, SAS Institute Inc., Cary, NC). Biomass of annual species was averaged across the thirty quadrats to obtain a representative value for each monitoring plot in each year. Similarly, representative values for perennial cover and density of seeded species were obtained by averaging over the three transects in each monitoring plot in each year. Prior to analysis, cover data were arcsine-square root transformed and density data were $\log_{10}$-transformed to meet the assumptions of equal variance and normality. The vegetation response variables (biomass, cover, density) were analyzed using mixed model procedures for repeated measures Analysis of Variance with plot as the experimental unit and replicate as a random factor. For each response variable, the most accurate covariance model structure for the repeated measures was selected based on comparisons using Akaike's Information Corrected Criterion (AICc), and type 3 tests of fixed effects (year, treatment and interaction) were then conducted using the selected model structure. Annual plant biomass was analyzed separately for native, non-native and seeded annual species. We also conducted separate analyses for each fire with seeding treatment as the main effect, monitoring plot as a blocking factor and the annual quadrat or perennial transect as the experimental unit. Separate analyses for the seed bank of seeded species were conducted for each year and for vacuumed (perennial species) versus cored (annual species) samples. Seed density of seeded perennial species was analyzed in separate analyses for each year (2006 and 2007) using negative binomial regression that included a Pearson estimate of over-dispersion for determining whether seeds of seeded species were more likely to occur on seeded versus unseeded burned areas (Sileshi 2007). Seed density of annual seeded species was similarly analyzed for the second year (2007). The proportions of recent desert tortoise sign (live tortoises, active burrows, fresh scat, and tracks) were compared across the treatments (untreated burn, seeded burn, and unburned) and all fires using Fisher's exact test.

RESULTS

Precipitation

Precipitation at the monitoring plots varied during the three-year period of monitoring (Figure 3-4). In 2005, only 1-2 mm fell at any site during winter (October - December) after seeding, far less than the amount considered vital for emergence of annual and early-season perennial species (Beatley 1974). Tramp was the only site where winter precipitation did not exceed 25 mm in 2006-07. In 2007-08 the Dry Middle, Dry Rock and Garnet fires received less than 14 mm (0.6") of winter precipitation. At Halfway and Meadow Valley in both 2006 and 2007, more than 56 mm (2.2") of precipitation fell during summer (May - September), a period coincident with emergence of late-season perennials. In 2007, Tramp and Bunkerville each received more than 38 mm (1.5") of summer precipitation. Although the Tramp Fire received the lowest rainfall of all sites during the important winter/spring period in 2005-

2006 and 2006-2007, this fire had double the precipitation compared to the Dry Rock, Dry Middle and Garnet fires in 2007-08, largely as a result of heavy rainfall in November 2007. Meadow Valley and Halfway fires, on the other hand, had the highest precipitation of all sites during winter/spring of 2005-2006 and 2006-2007, but received less rainfall than Tramp in 2007-2008, when the November 2007 storms largely missed these more northern sites.

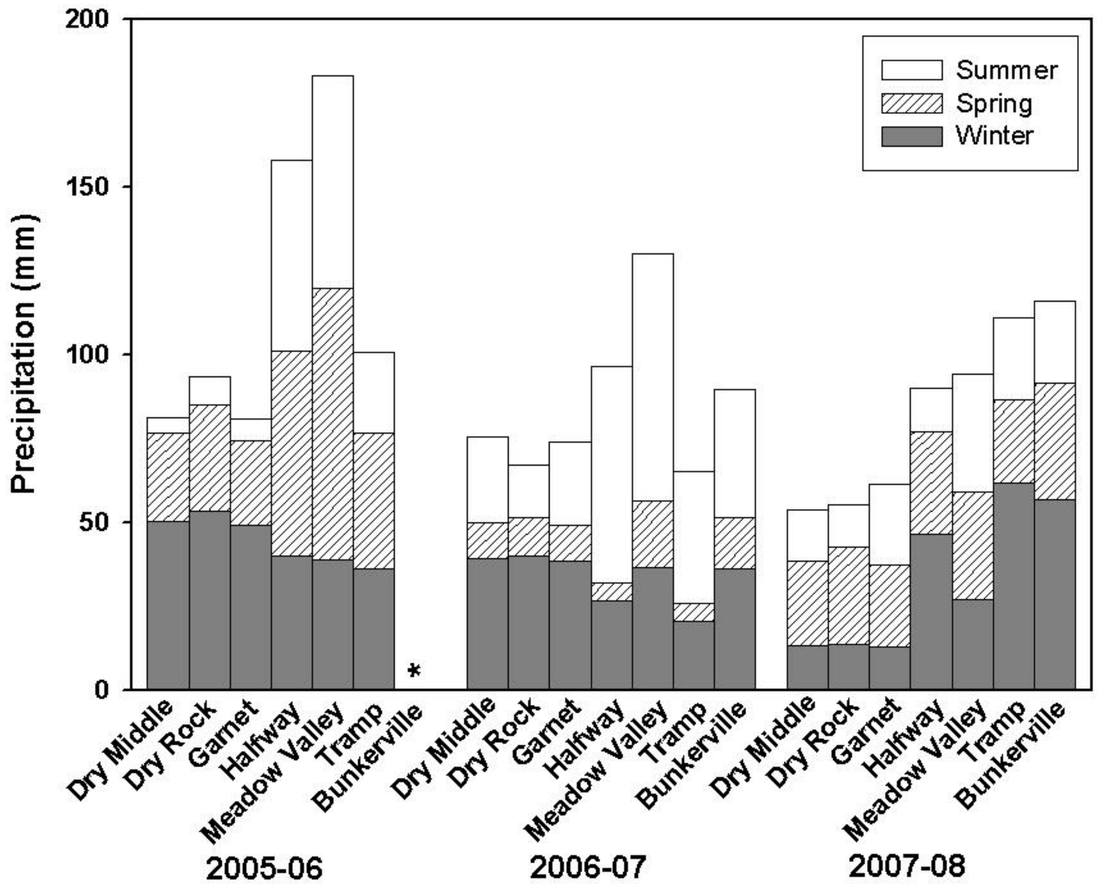

Figure 3-4. Precipitation for the hydrologic year (October through September) for three years after the 2005 fires were seeded. Winter precipitation (October – December) is shaded grey, spring precipitation (January - April) is hatched, and summer precipitation (May – September) is clear. Each bar represents the mean of all rain gauges placed within that fire. For months prior to rain gauge placement at the monitoring plots, data were reconstructed using the Parameter-elevation Regressions on Independent Slopes Model (PRISM) climate mapping system (PRISM Climate Group, Oregon State University; http://prism.oregonstate.edu/). Precipitation at the Bunkerville Fire is not shown for the 2005-06 season (*), as no seeding occurred at this fire until November, 2006.

Native, Non-Native and Needed Annual Biomass

Annual plant production varied over the three years following the Emergency-Stabilization seeding effort but was unaffected by the seeding treatment. Annual plant biomass was positively correlated with total precipitation from October through April (Figure 3-5), with no difference between the seeded and unseeded burn treatments. Over all fires, annual biomass was higher in 2006 compared with 2007 and 2008 for native ($F_{2,31.1} = 10.8$, $P <$ 0.01) and non-native species ($F_{2,76.9} = 76.4$, $P < 0.01$). Over the three years, native and non-native biomasses were equivalent on seeded and unseeded burn plots (Tables 3-2 and 3-3). Annual biomass of the seeded species was generally low among the fires, but was higher in 2008 compared to 2007 ($F_{1,48} = 30.95$, $P < 0.01$). As with the other annual species, production of seeded species was equal on seeded and unseeded burn plots (Table 3-4).

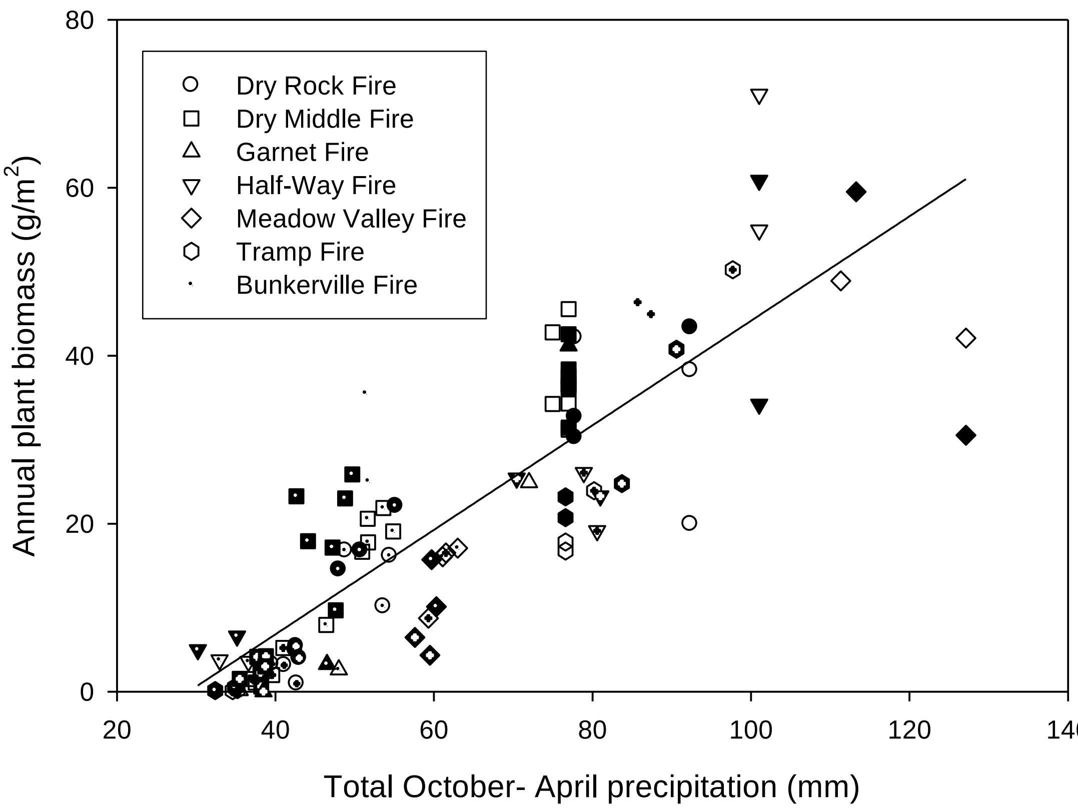

Figure 3-5. Average annual plant shoot biomass as a function of total winter/spring (October – April) precipitation ($F_{1,98} = 256.1$, $P < 0.01$; $r^2 = 0.72$). Symbols are coded by fire (see Legend). Solid symbols represent the seeded-burned plots, while clear symbols represent the unseeded-burned plots. Plain symbols indicate 2006 shoot biomass, symbols with an interior dot indicate 2007, and symbols with an interior cross-hair indicate 2008. Overall regression is annual biomass (g/m^2) = 0.62 * Oct-May precipitation (mm) – 18.06.

Table 3-2. Mean spring shoot biomass of non-native annual species (g/m^2 ± SE) across replicated quadrats within each monitoring plot at each fire. Treatment combinations marked with * were not sampled in 2006. The unburned values were not included in statistical analysis, and are shown solely for reference. Plots with biomass less than 0.1 g/m^2 are denoted by "0.0 ± 0.0"; those completely devoid of plants are denoted as "no plants". See text for statistical analysis of within-fire comparisons.

		2006			2007			2008		
Fire	Block	Burned	Seeded	Unburned	Burned	Seeded	Unburned	Burned	Seeded	Unburned
Bunkerville	B1	*	*	*	23.8 ± 2.0	33.5 ± 3.8	11.1 ± 1.8	37.2 ± 4.7	42.5 ± 7.7	12.2 ± 1.8
Dry Middle	CS1	28.6 ± 2.5	31.3 ± 4.1	*	4.5 ± 1.0	6.0 ± 1.1	1.1 ± 0.4	2.2 ± 0.5	0.5 ± 0.2	no plants
	CS3	32.8 ± 3.6	30.3 ± 3.7	9.4 ± 1.1	15.7 ± 2.2	17.1 ± 2.5	2.0 ± 0.4	1.0 ± 0.5	1.5 ± 0.4	0.1 ± 0.1
	CS4	30.5 ± 4.1	34.2 ± 3.5	*	17.1 ± 2.4	24.7 ± 3.9	1.6 ± 0.4	1.5 ± 0.4	4.1 ± 0.9	0.0 ± 0.0
	CS5	40.6 ± 5.4	36.1 ± 3.4	*	18.9 ± 2.2	22.5 ± 2.8	2.0 ± 0.5	5.2 ± 1.0	3.9 ± 0.9	0.1 ± 0.0
	CS6	37.0 ± 3.3	38.3 ± 3.7	*	21.1 ± 2.4	22.9 ± 2.7	2.8 ± 0.5	1.9 ± 0.6	2.3 ± 0.6	0.1 ± 0.0
	CS7	37.2 ± 3.3	38.2 ± 4.0	9.6 ± 1.6	17.6 ± 3.2	13.4 ± 2.5	1.7 ± 0.3	2.4 ± 0.6	2.9 ± 0.5	0.9 ± 0.2
Dry Rock	CS10	39.5 ± 3.2	29.7 ± 3.3	*	15.4 ± 1.9	13.7 ± 2.5	1.5 ± 0.4	3.1 ± 0.6	4.9 ± 0.8	0.2 ± 0.1
	CS8	37.5 ± 5.1	40.9 ± 4.4	*	15.6 ± 2.4	21.3 ± 3.7	1.5 ± 0.3	3.3 ± 0.6	5.4 ± 0.7	0.3 ± 0.2
	CS9	17.1 ± 3.5	31.5 ± 4.0	9.2 ± 1.6	9.3 ± 1.8	16.4 ± 3.6	2.1 ± 0.4	1.0 ± 0.3	3.9 ± 0.6	0.6 ± 0.4
Garnet	CS2	21.3 ± 1.9	34.8 ± 3.6	0.9 ± 0.3	1.8 ± 0.2	3.0 ± 0.6	0.8 ± 0.2	0.1 ± 0.1	0.0 ± 0.0	no plants
Halfway	H1	47.2 ± 6.2	29.7 ± 3.5	15.0 ± 2.3	1.4 ± 0.5	4.9 ± 0.9	1.4 ± 0.4	14.2 ± 1.6	15.7 ± 2.3	6.9 ± 1.1
	H2	55.5 ± 5.9	58.8 ± 6.9	19.6 ± 2.5	1.9 ± 0.4	4.6 ± 1.5	0.7 ± 0.2	21.0 ± 2.7	21.6 ± 2.7	9.2 ± 1.7
Meadow Valley	MV1	48.1 ± 5.2	27.6 ± 5.9	17.0 ± 3.3	11.2 ± 1.7	11.6 ± 2.0	3.5 ± 0.7	15.0 ± 2.0	4.0 ± 0.7	2.1 ± 0.6
	MV2	34.5 ± 5.7	46.0 ± 6.0	*	15.9 ± 2.0	9.8 ± 1.2	2.3 ± 0.6	7.7 ± 1.1	5.8 ± 1.1	2.9 ± 0.8
Tramp	T1	10.0 ± 2.8	1.9 ± 0.5	6.6 ± 2.0	0.0 ± 0.0	0.0 ± 0.0	0.2 ± 0.1	10.8 ± 2.0	10.4 ± 1.5	10.2 ± 1.8
	T2	10.1 ± 2.7	6.1 ± 1.7	4.3 ± 2.0	0.2 ± 0.1	0.1 ± 0.0	0.0 ± 0.0	19.8 ± 4.6	13.8 ± 2.0	4.1 ± 1.4

Table 3-3. Mean spring shoot biomass of native annual species (g/m^2 ± SE) across replicated quadrats within each monitoring plot at each fire. Treatment combinations marked with * were not sampled in 2006. The unburned values were not included in statistical analysis, and are shown solely for reference. Plots with biomass less than 0.1 g/m^2 are denoted by "0.0 ± 0.0"; those completely devoid of plants are denoted as "no plants". See text for statistical analysis of within-fire comparisons

		2006			2007			2008		
Fire	Block	Burned	Seeded	Unburned	Burned	Seeded	Unburned	Burned	Seeded	Unburned
Bunkerville	B1	*	*	*	0.7 ± 0.3	2.0 ± 0.6	0.8 ± 0.3	7.7 ± 1.7	3.8 ± 0.6	14.4 ± 2.0
Dry Middle	CS1	5.7 ± 1.0	4.7 ± 1.3	*	1.2 ± 0.3	1.9 ± 0.4	0.1 ± 0.0	0.1 ± 0.0	0.0 ± 0.1	0.1 ± 0.0
	CS3	1.4 ± 0.3	1.1 ± 0.5	1.6 ± 0.3	1.0 ± 0.3	0.8 ± 0.4	0.3 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	0.3 ± 0.2
	CS4	0.7 ± 0.4	2.2 ± 0.7	*	0.6 ± 0.3	1.2 ± 0.5	0.4 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.1
	CS5	2.2 ± 0.9	0.5 ± 0.3	*	0.2 ± 0.1	0.5 ± 0.2	0.3 ± 0.1	0.0 ± 0.0	0.3 ± 0.2	0.0 ± 0.0
	CS6	0.5 ± 0.2	0.1 ± 0.1	*	0.8 ± 0.4	0.3 ± 0.2	0.1 ± 0.1	0.1 ± 0.1	no plants	0.0 ± 0.0
	CS7	8.4 ± 2.2	4.3 ± 0.7	0.5 ± 0.2	2.9 ± 0.9	3.8 ± 0.9	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.1	0.0 ± 0.0
Dry Rock	CS10	2.7 ± 0.7	0.5 ± 0.2	*	1.4 ± 0.7	0.8 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.0	0.0 ± 0.0
	CS8	0.8 ± 0.3	2.4 ± 1.7	*	0.5 ± 0.5	0.7 ± 0.4	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	CS9	2.9 ± 1.2	1.2 ± 0.6	0.4 ± 0.1	0.9 ± 0.3	0.4 ± 0.2	0.1 ± 0.0	no plants	0.0 ± 0.0	0.2 ± 0.1
Garnet	CS2	3.6 ± 0.8	6.4 ± 2.1	0.1 ± 0.1	0.4 ± 0.1	0.2 ± 0.1	0.2 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	no plants
Halfway	H1	7.7 ± 1.7	4.5 ± 1.2	3.5 ± 1.0	1.7 ± 0.4	0.2 ± 0.1	0.1 ± 0.0	5.0 ± 0.9	7.6 ± 2.0	5.3 ± 1.0
	H2	15.6 ± 4.9	2.1 ± 0.8	4.3 ± 1.6	1.8 ± 0.6	0.3 ± 0.2	0.01 ± 0.0	5.1 ± 1.0	3.8 ± 0.6	7.5 ± 2.1
Meadow Valley	MV1	0.8 ± 0.3	3.0 ± 0.8	0.9 ± 0.2	0.2 ± 0.1	0.1 ± 0.0	0.4 ± 0.3	1.5 ± 0.3	0.4 ± 0.2	0.7 ± 0.1
	MV2	7.6 ± 4.5	13.5 ± 4.7	*	0.2 ± 0.01	0.3 ± 0.1	0.2 ± 0.1	1.0 ± 0.5	0.7 ± 0.2	1.6 ± 0.4
Tramp	T1	7.8 ± 1.3	21.3 ± 3.2	8.3 ± 1.0	0.1 ± 0.0	0.4 ± 0.2	0.0 ± 0.0	13.1 ± 1.1	14.3 ± 2.0	17.2 ± 1.1
	T2	6.7 ± 1.4	14.6 ± 1.8	4.3 ± 1.0	0.0 ± 0.0	0.0 ± 0.0	no plants	30.4 ± 4.1	26.9 ± 2.8	14.7 ± 2.2

Table 3-4. Mean spring shoot biomass of seeded annual species (g/m^2 ± SE) across replicated quadrats within each monitoring plot at each fire. As seeding with annual species had not yet occurred, seeded annual species were not sampled separate of other native species in 2006. The unburned values were not included in statistical analysis, and are shown solely for reference. Plots with biomass less than 0.1 g/m^2 are denoted by "0.0 ± 0.0"; those completely devoid of plants are denoted as "no plants". No within-fire comparisons of seeded and unseeded burn plots were statistically different in 2006, 2007 or 2008.

			2007			2008	
Fire	Block	Burned	Seeded	Unburned	Burned	Seeded	Unburned
Bunkerville	B1	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.1	0.7 ± 0.2	0.4 ± 0.1	2.9 ± 0.6
Dry Middle	CS1	no plants	no plants	no plants	no plants	0.0 ± 0.0	no plants
	CS3	no plants	no plants	no plants	no plants	no plants	0.0 ± 0.0
	CS4	no plants	no plants	no plants	no plants	no plants	no plants
	CS5	0.0 ± 0.0	no plants	no plants	no plants	no plants	no plants
	CS6	no plants	0.0 ± 0.0	0.0 ± 0.0	no plants	no plants	no plants
	CS7	no plants	no plants	no plants	no plants	0.0 ± 0.0	0.0 ± 0.0
Dry Rock	CS10	no plants	no plants	no plants	no plants	0.0 ± 0.0	0.0 ± 0.0
	CS8	no plants	no plants	no plants	0.0 ± 0.0	0.0 ± 0.0	no plants
	CS9	no plants	no plants	no plants	no plants	0.0 ± 0.0	0.0 ± 0.0
Garnet	CS2	no plants	no plants	no plants	0.0 ± 0.0	no plants	no plants
Halfway	H1	no plants	no plants	no plants	no plants	0.2 ± 0.1	0.2 ± 0.1
	H2	no plants	no plants	no plants	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.0
Meadow Valley	MV1	no plants	no plants	no plants	0.1 ± 0.0	0.1 ± 0.1	0.2 ± 0.1
	MV2	0.0 ± 0.0	0.0 ± 0.0	no plants	0.1 ± 0.1	0.1 ± 0.0	0.3 ± 0.1
Tramp	T1	no plants	no plants	no plants	0.4 ± 0.1	0.1 ± 0.0	1.6 ± 0.5
	T2	no plants	no plants	no plants	0.2 ± 0.1	0.7 ± 0.2	0.8 ± 0.3

Within individual fires and for the individual years, the differences in annual plant production between seeded and unseeded plots were not predictable. In 2006, native annual production was double on seeded compared with unseeded-burn plots within the Meadow Valley ($F_{2,146}$ = 8.3, P < 0.01) and Tramp fires ($F_{2,175}$ = 15.9, P < 0.01), but lower on seeded than unseeded-burn plots at the Halfway Fire ($F_{2,177}$ = 10.7, P < 0.01; Table 3-3). The Tramp Fire also had a more than two-fold reduction in non-native annual plant production in 2006 on seeded compared with unseeded-burn plots ($F_{2,176}$= 3.9, P = 0.02), while non-native biomass increased on seeded compared with unseeded-burn plots at the Garnet Fire ($F_{2,167}$ = 87.7, P < 0.01; Table 3-2). In 2007, the seeded plots had lower native annual production ($F_{2,164}$ = 20.9, P < 0.01) and higher non-native production ($F_{2,165}$ = 19.6, P < 0.01) than unseeded-burn plots at the Halfway Fire (Tables 3-2 and 3-3). In 2008, native annual biomass was lower in seeded plots compared with unseeded-burn plots at the Meadow Valley Fire ($F_{1,117}$ = 4.6, P = 0.03; Table 3-3). Non-native annual plant production in 2008 was lower in seeded plots compared with unseeded-burn plots at the Meadow Valley ($F_{2,177}$ = 36.1, P < 0.01) and Garnet fires ($F_{2,87}$ = 7.8, P < 0.01), but higher in the seeded compared with unseeded-burn plots at the Dry Rock Fire ($F_{2,257}$ = 79.0, P < 0.01; Table 3-2). All other within-fire comparisons were not significantly different at the 0.05 level.

Perennial Plant Cover and Seedling Density

Live perennial plant cover across all fires increased slightly between 2006 and 2007, leveling off in 2008 ($F_{1,78.2}$ = 6.7, *P* < 0.01; Figure 3-6A). Throughout the three years of monitoring, live perennial cover was equal on seeded and untreated burn plots (Figure 3-6A, Table 3-5). Appendix 3-1 details the species composition of live cover on the plots. Seedling density of seeded perennial species, on the other hand, was initially similar between seeded and untreated burn plots in 2006, but increased on the seeded plots in 2007 and remained higher in 2008 (Year × Treatment interaction, $F_{2,31}$ = 5.0, *P* = 0.01; Figure 3-6B, Table 3-6).

Within-fire comparisons in 2007 demonstrated higher mean densities in seeded plots compared with unseeded burn plots at the Meadow Valley Fire ($F_{2,14}$ = 30.4, *P* < 0.01). Likewise, within-fire comparisons in 2008 demonstrated higher mean densities in seeded plots compared with unseeded burn plots at the Meadow Valley ($F_{2,14}$ = 13.76, *P* < 0.01), Garnet ($F_{2,6}$ = 12.7, *P* < 0.01), and Bunkerville fires ($F_{2,6}$ = 7.36, *P* = 0.02). In both years, this increase was primarily due to desert globemallow (*Sphaeralcea ambigua*) and desert marigold (*Baileya multiradiata*).

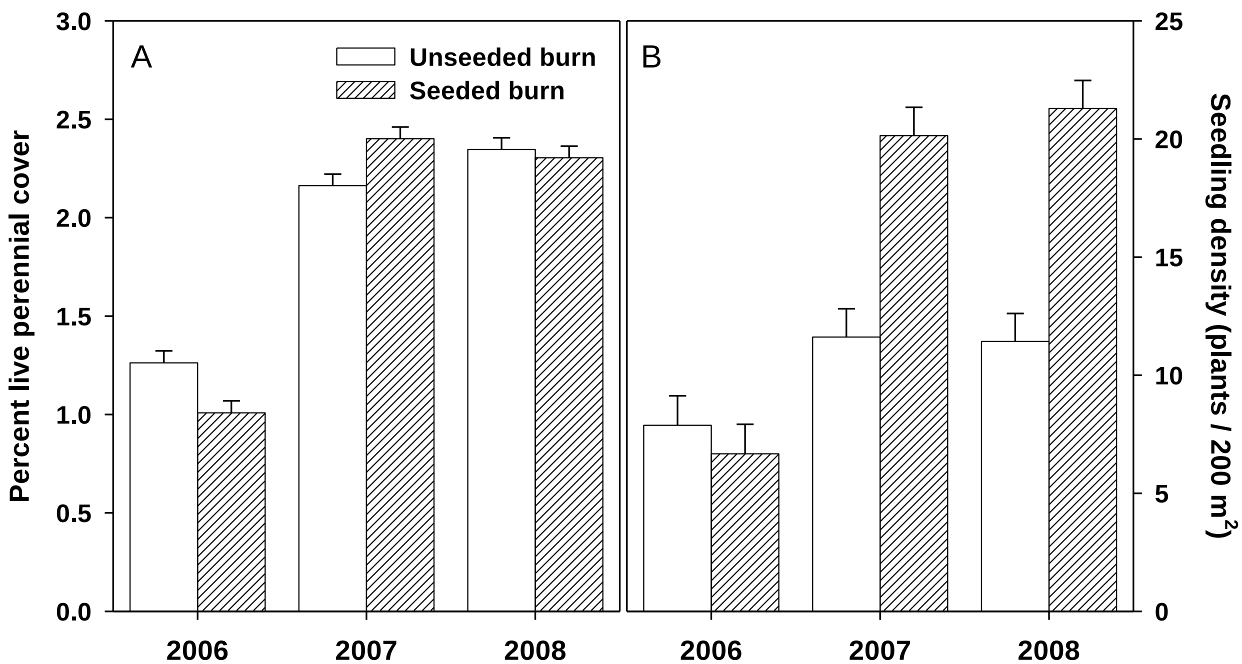

Figure 3-6. Live perennial plant cover (A) and seedling density of seeded perennial species (B) for the unseeded burn and seeded burn plots in 2006, 2007 and 2008. Means and standard errors presented are back-transformed values from statistical analyses.

Table 3-5. Mean percent live perennial plant cover (± SE) across transects established for each monitoring plot within fire. Plots at Bunkerville were established in 2007. The unburned values were not included in statistical analysis, and are shown solely for reference. Plots with canopy cover less than 0.1 % are denoted by "0.0 ± 0.0"; those completely devoid of plants denoted as "no plants". No within-fire comparisons of seeded and unseeded burn plots were statistically different in 2006, 2007 or 2008.

			2006			2007			2008	
Fire	Block	Burned	Seeded	Unburned	Burned	Seeded	Unburned	Burned	Seeded	Unburned
Bunkerville	B1	*	*	*	1.6 ± 0.9	1.4 ± 0.7	15.2 ± 1.8	1.6 ± 0.9	3.0 ± 0.7	15.5 ± 2.5
Dry Middle	CS1	3.2 ± 3.2	no plants	29.7 ± 3.0	0.0 ± 0.0	3.9 ± 3.9	29.3 ± 0.7	no plants	3.6 ± 3.4	22.3 ± 5.3
	CS3	no plants	0.2 ± 0.2	30.9 ± 7.0	1.0 ± 0.9	1.8 ± 1.0	30.0 ± 4.3	0.6 ± 0.2	1.8 ± 0.4	23.6 ± 2.9
	CS4	0.2 ± 0.1	0.0 ± 0.0	31.4 ± 5.0	2.8 ± 0.9	2.1 ± 1.3	28.1 ± 3.4	1.4 ± 0.4	0.8 ± 0.3	23.6 ± 2.3
	CS5	0.2 ± 0.1	0.3 ± 0.1	33.2 ± 2.9	3.2 ± 2.0	1.9 ± 1.5	27.0 ± 2.2	4.5 ± 1.2	1.3 ± 0.5	23.5 ± 2.8
	CS6	0.1 ± 0.1	0.0 ± 0.0	22.4 ± 1.6	1.9 ± 0.5	1.3 ± 0.6	25.7 ± 4.5	2.6 ± 1.2	2.2 ± 0.7	21.2 ± 2.4
	CS7	0.4 ± 0.4	0.1 ± 0.1	24.8 ± 1.3	3.1 ± 2.6	0.5 ± 0.3	31.7 ± 4.5	1.3 ± 0.5	0.2 ± 0.1	28.7 ± 6.8
Dry Rock	CS10	1.6 ± 0.4	no plants	20.5 ± 6.3	3.1 ± 1.9	0.7 ± 0.6	19.0 ± 2.0	3.7 ± 3.7	0.4 ± 0.1	16.9 ± 4.4
	CS8	0.6 ± 0.6	0.0 ± 0.0	23.9 ± 3.9	3.1 ± 2.0	1.3 ± 0.7	19.0 ± 2.6	1.6 ± 1.6	0.3 ± 0.2	21.9 ± 1.9
	CS9	2.0 ± 0.9	0.1 ± 0.1	21.3 ± 3.3	2.4 ± 0.9	3.4 ± 1.1	17.6 ± 1.6	2.5 ± 0.8	1.0 ± 0.9	15.9 ± 0.7
Garnet	CS2	0.5 ± 0.2	0.0 ± 0.0	18.4 ± 3.0	0.7 ± 0.4	0.5 ± 0.3	15.1 ± 1.8	0.7 ± 0.4	0.2 ± 0.1	14.0 ± 0.8
Halfway	H1	3.3 ± 1.3	8.2 ± 1.2	18.9 ± 2.6	2.4 ± 0.2	6.1 ± 1.8	15.7 ± 2.1	4.5 ± 0.5	5.6 ± 0.3	13.9 ± 1.2
	H2	0.4 ± 0.3	1.4 ± 0.7	16.5 ± 2.7	1.0 ± 0.4	1.3 ± 0.2	14.3 ± 0.3	1.6 ± 0.8	1.5 ± 0.9	9.0 ± 0.8
Meadow Valley	MV1	9.3 ± 3.7	18.5 ± 1.6	31.8 ± 3.0	8.8 ± 1.9	14.2 ± 1.8	20.4 ± 2.3	13.0 ± 2.0	15.2 ± 3.2	28.2 ± 3.6
	MV2	6.8 ± 3.2	2.1 ± 0.8	30.3 ± 2.9	10.3 ± 2.7	10.1 ± 4.7	18.1 ± 2.3	11.5 ± 3.4	12.6 ± 1.9	25.7 ± 1.0
Tramp	T1	4.8 ± 4.1	8.5 ± 3.6	16.2 ± 2.6	0.9 ± 0.4	2.7 ± 1.2	11.8 ± 0.9	1.7 ± 0.9	4.5 ± 1.4	13.2 ± 2.3
	T2	no plants	1.6 ± 0.2	15.6 ± 1.9	0.1 ± 0.1	0.2 ± 0.1	9.6 ± 0.4	0.2 ± 0.1	1.0 ± 0.4	11.4 ± 0.9

Table 3-6. Mean seeded perennial plant density (plants/200m^2 ± SE) across transects established for each monitoring plot within fire. Plots at Bunkerville were established in 2007. The unburned values were not included in statistical analysis, and are shown solely for reference. Plots with no seeded plants are denoted as "no plants". See text for statistical analysis of within-fire comparisons.

		2006			2007			2008		
Fire	Block	Burned	Seeded	Unburned	Burned	Seeded	Unburned	Burned	Seeded	Unburned
Bunkerville	B1	*	*	*	11.0 ± 2.1	31.0 ± 13.7	1.7 ± 1.2	14.3 ± 4.4	42.3 ± 10.2	7.7 ± 3.8
Dry Middle	CS1	0.3 ± 0.3	0.7 ± 0.7	no plants	1.0 ± 0	4.7 ± 1.2	no plants	1.0 ± 0	6.7 ± 1.5	no plants
	CS3	12 ± 4.2	45.7 ± 26.0	1.7 ± 1.7	17.0 ± 6.8	69.0 ± 22.2	0.3 ± 0.3	14.7 ± 6.1	74.3 ± 13.0	0.3 ± 0.3
	CS4	42.7 ± 20.8	11.7 ± 5.9	1.3 ± 0.9	49.7 ± 16.4	38.7 ± 11.7	no plants	44.0 ± 6.2	43.0 ± 10.1	no plants
	CS5	162.7 ± 69.3	82.3 ± 46.6	2.0 ± 2.0	158.7 ± 69.2	130.3 ± 49.3	0.7 ± 0.3	140.0 ± 57.4	138.0 ± 47.6	1.0 ± 0.6
	CS6	42.7 ± 3.5	27.0 ± 12.8	0.3 ± 0.3	63.7 ± 6.2	69.3 ± 14.2	no plants	66.0 ± 3.8	82.7 ± 21.2	no plants
	CS7	7.3 ± 3.3	3.0 ± 2.5	no plants	17.7 ± 7.5	20.0 ± 2.3	no plants	15.0 ± 4.9	9.0 ± 2.6	no plants
Dry Rock	CS10	6.7 ± 4.4	4.0 ± 2.1	1.0 ± 0.6	5.0 ± 1.5	47.3 ± 17.6	6.0 ± 5.5	9.0 ± 7.0	23.3 ± 9.0	7.0 ± 5.6
	CS8	14.7 ± 5.9	8.3 ± 2.7	0.3 ± 0.3	11.0 ± 3.5	17.0 ± 2.1	no plants	11.7 ± 3.8	24.7 ± 12.3	no plants
	CS9	38.3 ± 10.7	10.0 ± 1.2	5.3 ± 3.9	35.0 ± 13.6	20.0 ± 4.0	5.3 ± 2.7	37.3 ± 13.8	13.7 ± 3.8	2.7 ± 1.5
Garnet	CS2	no plants	3.7 ± 1.9	no plants	2.3 ± 2.3	19.3 ± 9.1	0.3 ± 0.3	1.7 ± 1.2	22.3 ± 5.9	no plants
Halfway	H1	10.0 ± 7.0	17.7 ± 7.1	5.5 ± 0.4	17.3 ± 8.4	34.0 ± 13.3	41.7 ± 35.3	16.7 ± 6.2	30.3 ± 7.0	11.3 ± 7.8
	H2	0.7 ± 0.7	no plants	10.3 ± 7.4	2.0 ± 0.6	13.3 ± 4.4	14.0 ± 6.6	3.0 ± 0.6	9.3 ± 3.0	13.7 ± 9.7
Meadow Valley	MV1	37.7 ± 14.3	39.0 ± 22.0	4.3 ± 3.8	50.3 ± 15.1	104.7 ± 24.3	5.3 ± 3.0	58.3 ± 14.7	88.7 ± 19.3	26.3 ± 18.6
	MV2	100.7 ± 20.7	127.7 ± 17.5	4.0 ± 2.6	89.3 ± 20.2	168.7 ± 15.2	6.3 ± 2.7	79.0 ± 21.5	156.3 ± 16.0	30.0 ± 8.9
Tramp	T1	no plants	no plants	0.7 ± 0.7	1.0 ± 1.0	no plants	no plants	no plants	0.7 ± 0.7	1.0 ± 0.6
	T2	1.0 ± 1.0	no plants	0.3 ± 0.3	0.3 ± 0.3	no plants	no plants	1.0 ± 0.6	0.3 ± 0.3	0.3 ± 0.3

Seeded Species in the Seed Bank

Although viable seed densities were much lower than initial seeding rates, treated areas retained seeds months after they were hand broadcast. In 2006, ten months after the first seed mix was applied at all fires except Bunkerville, the overall seed density of the applied perennial species was more than six times higher in seeded areas than in areas without seeding (df = 1, Chi-square = 46.37, $P < 0.01$; Table 3-7). The greater seed densities in treated areas were mainly due to retention of the perennial herb desert marigold. In 2007, eleven months after the second seed application, the density of applied perennial seed species remained more then four times higher in seeded areas compared with controls (df = 1, χ^2 = 22.01, $P < 0.01$), due to augmentation of desert marigold, bursage, and cheesebush (*Hymenoclea salsola*; Table 3-7). Interestingly, desert globemallow seed densities increased between 2006 and 2007 in areas where seed was not applied. In contrast to perennial seeded species, viable seed densities of applied annual species were not different in seeded and unseeded burn plots in 2007 (Table 3-7).

Tortoise Activity

Tortoise activity was not significantly different between seeded and untreated burned areas during the three years of monitoring. Table 3-8 details all tortoise sign found within the plots, while only recent sign (live tortoises, active burrows, fresh scat, and tracks) were used in the analyses. Proportionally less recent tortoise sign occurred in burned areas, regardless of seeding or no seeding, compared with adjacent unburned areas in 2006 (df = 6, n = 275, $P < 0.01$), 2007 (df = 6, n = 284, $P = 0.04$), and 2008 (df = 6, n = 535, $P < 0.01$). Even when unburned areas were removed from the analyses, proportions of tortoise sign were not significantly different between seeded and unseeded burned areas.

Table 3-7. Densities (mean ± SD) of seeded perennial and annuals species remaining in seeded and unseeded burned surface soils. Perennials in the seed bank represent one year (2006) and two years (2007) following the initial seeding in December 2005. Annuals represent one year (2007) after annals were seeded in November 2006.

Perennial seeds/4 m^2 in 2006		
	Control	Seeded
B. multiradiata	0.08±0.38	1.81±4.11
A. dumosa	0.22±0.69	0.36±0.81
S. ambigua	0.01±0.08	0.00±0.00
A. canescens	0.00±0.00	0.01±0.12
Total	0.31±0.77	2.19±4.31

Perennial seeds/4 m^2 in 2007		
	Control	Seeded
B. multiradiata	0.31±0.29	2.84±4.74
A. dumosa	0.18±0.52	0.46±0.85
S. ambigua	0.78±3.91	0.72±1.62
A. canescens	0.00±0.00	0.24±0.71
A. hymenoides	0.00±0.00	0.02±0.15
E. virgenensis	0.01±0.11	0.07±0.30
H. salsola	0.01±0.11	0.46±1.03
L. tridentata	0.01±0.11	0.01±0.11
Total	1.29±4.29	4.81±6.77

Annual seeds/m^2 in 2007		
	Control	Seeded
V. octoflora	51.2±87.6	49.9±91.6
M. glabrata	2.8±10.1	6.0±17.8
B. aristidoides	1.4±7.8	0.3±3.3
C. claviformis	0.4±3.3	0.0±0.0
Total	55.7±89.6	55.8±96.5

Table 3-8. Summary of desert tortoise sign recorded for untreated burn (B), seeded burn (S), and unburned reference sites (U) during three years of monitoring in the Southern Nevada Complex.

		Bunkerville			Dry Middle			Dry Rock			Garnet			Halfway			Meadow Vy			Tramp			Total
	Sign	B	S	U	B	S	U	B	S	U	B	S	U	B	S	U	B	S	U	B	S	U	
2006	Burrow				5	14	20	19	8	19	0	2	2	12	20	10	3	1	6	24	17	45	**227**
	Carcass				3	4	4	2	7	10	1	0	0	0	3	0	0	0	0	2	0	1	**37**
	Eggshells				0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	**1**
	Fresh Scat				0	0	6	0	0	8	0	0	2	0	0	0	0	0	0	2	0	8	**26**
	Tortoise				0	0	2	3	0	2	0	0	1	0	0	1	0	0	0	1	1	4	**15**
	Tracks				0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	3	1	2	**7**
	Total				*8*	*19*	*34*	*28*	*15*	*56*	*1*	*2*	*5*	*12*	*23*	*11*	*3*	*1*	*6*	*42*	*20*	*65*	***313***
2007	Burrow	1	0	4	12	10	18	21	18	26	3	1	2	5	4	3	0	0	1	28	27	46	**230**
	Carcass	0	0	0	5	2	5	8	10	12	1	0	0	1	3	0	0	0	0	3	1	4	**55**
	Fresh Scat	0	0	0	0	2	5	2	3	16	1	0	0	2	0	1	0	0	1	0	0	1	**34**
	Tortoise	0	0	0	1	1	3	1	1	6	0	0	0	0	0	1	0	0	0	0	0	1	**15**
	Tracks	0	0	0	0	0	0	1	1	1	0	0	0	0	0	1	0	0	0	0	0	1	**5**
	Total	*1*	*0*	*4*	*18*	*18*	*32*	*33*	*33*	*75*	*5*	*1*	*3*	*8*	*7*	*6*	*0*	*0*	*2*	*33*	*29*	*57*	***339***
2008	Burrow	5	2	1	43	29	57	51	27	67	17	18	4	22	32	26	3	7	5	21	21	36	**494**
	Carcass	0	0	0	9	2	8	6	12	16	1	0	0	1	2	1	0	0	1	2	1	1	**63**
	Fresh Scat	0	0	0	0	1	7	2	1	9	0	0	0	1	0	0	0	0	0	1	2	4	**28**
	Tortoise	0	0	0	0	2	1	1	2	1	0	0	0	0	0	0	0	0	1	0	1	1	**10**
	Tracks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	2	**3**
	Total	*5*	*2*	*1*	*52*	*34*	*73*	*61*	*45*	*104*	*18*	*18*	*5*	*25*	*34*	*28*	*3*	*8*	*7*	*25*	*25*	*45*	***598***

DISCUSSION

The 2005 BAER plan determined that the burn severity of desert tortoise Critical Habitat in the Southern Nevada Fire Complex warranted intervention to protect tortoise populations. The primary objective of this study was to determine the effectiveness of seeding burned areas with native Mojave Desert annual and perennial species to rehabilitate desert tortoise habitat and restore tortoise activity in the burned areas. Determining if and how tortoises use burned habitat and whether habitat can be restored rapidly is critical to recovery of the species as habitat continues to be altered by wildfires and other disturbances. Tortoise activity, as indicated by tortoise sign, was not enhanced in seeded areas during the three years following seeding. Tortoise sign was found more frequently in unburned areas than burned areas overall, a result supported by observations of radio-telemetered tortoises in the Dry Middle, Dry Rock and Garnet fires (K. Drake, unpublished data). In this companion study, tortoises were observed moving along fire boundaries, using vegetation and burrows as shelter in unburned habitat while foraging and basking within burned areas where more herbaceous forage is available.

Annual plant production during the three years of monitoring consisted primarily of non-native brome grass, Mediterranean grass, and filaree (*Erodium cicutarium*). While total annual plant production was positively correlated to October-April precipitation, there was no consistent pattern in native versus non-native annual production. Successful establishment of seeded native perennial species likewise did not result in a decline of non-native annuals. A seeding study in the Sonoran Desert also found that, in spite of establishment of five seeded native species, non-native Mediterranean grass populations reached similar levels among seeded burned, unseeded burned, and unburned sites within 32 months (Abella *et al.* 2009). Seed bank studies conducted in the northeastern Mojave Desert revealed that Mediterranean grass and filaree experienced low seed mortality during experimental burns, while brome grass seeds experienced high mortality, especially under shrub canopies where fire temperatures were highest (Esque 2004). Filaree seeds employ active mechanisms that thrust seeds beneath the soil surface (Stamp 1984), while the seeds of Mediterranean grass, approximately the size of fine grains of sand, fall into cracks between soil particles (Gutterman 1994). The seeds of both species are thus able to escape lethal fire temperatures 1-2 cm below ground, where peak temperatures and residence time are much less than at or above the soil surface (Esque 2004). Although brome grass suffered high seed mortality during fire, the increased soil nitrogen that often accompanies wildfires in arid shrublands can boost non-native grass production during years of high rainfall (Hunter and Omi 2006, Esque 2004, Brooks 2002), allowing brome grasses to rapidly rebuild their seed bank. The consistently high production of non-native annual species in burned areas in this study suggests that, while precipitation is a primary limiting factor, native plant production has not been effective by itself to reduce non-native populations.

Regardless of non-native annual abundance, seeded areas contained higher seed bank densities of seeded perennial species compared with unseeded burn areas over all fires. By June 2007, seeded perennials had emerged on the seeded plots, except at the Tramp Fire and one plot at the Halfway Fire. Over all fires, the seedling density of seeded species was higher on seeded plots compared with unseeded burn plots in both the second and third years following seeding, contrasting with consistently low perennial cover in all three years. Perennial canopy cover is a common metric used to evaluate habitat recovery, but mounting evidence shows that seeding does not effectively improve short-term plant cover after wildfires (Robichaud *et al*. 2000, Beyers 2004, Robichaud *et al.* 2006, Dodson and Peterson 2009). Perennial seedling emergence and seed bank densities are likely more sensitive short-term measurements of seeding success but have been infrequently measured in post-fire monitoring. Failure of perennial species to establish at Tramp and Halfway, in spite of high seed bank densities, demonstrates that other site conditions, such as soils or site-specific germination temperatures, need to be considered when selecting species for broadcast seeding.

Seedling emergence on seeded areas consisted primarily of disturbance-tolerant native perennials in the seed mixture, such as desert globemallow and desert marigold. Five of the perennial species present in our seed mix (desert globemallow, desert marigold, cheesebush, bursage, and brittlebush (*Encelia virginensis*)) are among the most abundant naturally-establishing perennials after wildfire and other surface disturbances in the Mojave Desert (Walker and Powell 1999, Abella *et al*.2009). Application rates of desert globemallow and desert marigold were more than 24 times higher than the other three species, which could explain why they established more successfully. Seeding rates for desert globemallow, desert marigold, and the annual needle grama (*Bouteloua aristidoides*) all exceeded 250 live seeds/m^2, while the remaining four annual and six perennial species were applied at less than 30 live seeds/m^2. Greater establishment occurred for the same native annual species applied in mid-October 2007 to smaller experimental plots within the seven fires: annuals applied at a rate 10 times higher than in this study resulted in a 1.6 times increase in biomass of seeded spring annuals compared to unseeded plots (L. A. DeFalco *et al*., unpublished data). Because it is difficult to obtain large quantities of native seeds from commercial sources, efforts to develop local native seed sources and to synchronize seeding with winter rainfall are essential for rehabilitating Mojave Desert shrublands (Beatley 1974).

Currently, composition of seed mixes is largely based on commercial availability, but selection of species to complement site-specific properties is probably a more important consideration to promote seeding success after wildfire. Species selection can be guided by pre-burn conditions if vegetation maps are available (Abella *et al*. 2009) or nearby unburned areas can be used as a guide. Four-wing saltbush (*Atriplex canescens*), for example, although

not found in any nearby unburned areas, was seeded in both 2005 and 2006 but failed to establish on any seeded plot. On the other hand, the native annual needle grama was seen growing on unseeded plots, although in low numbers in small areas. Needle grama is a summer annual that requires abundant rainfall coupled with warm temperatures for germination (Went 1948), a combination that arises infrequently in the Mojave Desert when isolated storm events occur during monsoon season. Even though needle grama had the highest application rate of any annual or perennial species (except at the Bunkerville Fire), production was equivalent on seeded and unseeded plots after monsoons delivered localized precipitation to most fires in July and August of 2007 (L. A. DeFalco *et al*., unpublished data). As a summer annual, needle grama was subjected to a longer residence time on soil surfaces than early-season species waiting for suitable germination conditions, and seed densities were already depleted to levels equivalent to unseeded areas eleven months after application. The longer seeds reside on the soil surface, the greater their vulnerability to seed predators, fungal decomposition, and photo-degradation. Different techniques have been suggested to reduce seed vulnerability by covering seeds with soil (Bainbridge 2007), but damage to soil structure from heavy machinery and the high labor cost of applying treatments by hand over large areas limit their potential effectiveness, particularly in desert tortoise critical habitat. Alternately, reducing seed residence time by scheduling seed application to coincide with seasonal precipitation could provide a successful means to minimize seed losses.

In summary, within the BAER standard three-year monitoring period we found that perennial seedling emergence and seed bank densities were sensitive indicators of potential habitat recovery following the 2005 wildfires. Seeding with disturbance-tolerant perennials produced rapid seedling emergence and establishment, and seeded perennials were more abundant in the seed bank of seeded areas than unseeded areas. This augmentation of the seed bank demonstrates that seeding increased the long-term recovery potential of seeded burn sites. Maximizing seeding rates and reducing seed residence time with seasonally-appropriate application may improve establishment of native annuals and fast-establishing perennials and deserves further research. Studies in nearby arid lands have found that seeding can accelerate habitat recovery after wildfire (Eiswerth and Shonkwiler 2006, Eiswerth *et al*. 2009), and we expect that our continued assessment of these monitoring plots will shed new light on the efficacy of seeding burned Mojave Desert habitat. Desert tortoise activity in burned areas during the three year monitoring period was likely hindered by lack of shelter sites, and longer-term monitoring of both canopy cover and tortoise activity is necessary to determine the effectiveness of seeding for tortoise habitat restoration. As regeneration of plant communities will likely be influenced by changes in precipitation and temperature patterns expected for the arid Southwest (McCabe *et al.* 2004, Seager *et al.* 2007), understanding the indirect effects on tortoise populations of altered forage and cover availability after wildfire is critical for management and recovery of this species.

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Appendix 3-1. Contribution by species to live perennial cover for unseeded-burn and seeded monitoring plots within fire. Plots at Bunkerville and the unseeded plot at MV2 were established in 2007. Contribution is given as the percent of live perennial plant cover provided by that species. Plants contributing < 0.1% are denoted as "0.0" and plants not present are denoted as "-".

			2006		2007		2008	
Fire	Block	Species	Burned	Seeded	Burned	Seeded	Burned	Seeded
Bunkerville	B1	*Ambrosia dumosa*	*	*	11.2	0.2	-	7.8
		Ephedra nevadensis	*	*	-	9.2	-	-
		Eriogonum inflatum	*	*	68.2	4.8	69.2	43.3
		Krameria species	*	*	-	16.4	10.9	1.2
		Larrea tridentata	*	*	-	-	1.0	-
		Lycium species	*	*	-	-	3.9	-
		Psorothamnus fremontii	*	*	3.6	-	2.5	13.5
		Sphaeralcea ambigua	*	*	1.3	7.2	12.5	28.2
		Thamnosma montana	*	*	15.6	-	-	-
		Yucca brevifolia	*	*	-	62.2	-	5.9
		Yucca schidigera	*	*	0.2	-	-	-
Dry Middle	CS1	*Acamptopappus sphaerocephalus*	11.0	-	-	7.1	-	3.8
		Ambrosia dumosa	29.6	-	100.0	40.5	-	33.6
		Coleogyne ramosissima	-	-	-	-	-	1.1
		Ephedra nevadensis	3.5	-	-	3.6	-	2.2
		Krameria species	3.1	-	-	-	-	-
		Larrea tridentata	38.0	-	-	29.7	-	40.6
		Sphaeralcea ambigua	-	-	-	-	-	4.0
		Yucca brevifolia	14.7	-	-	11.8	-	6.1
		Yucca schidigera	0.1	-	-	7.3	-	8.4
	CS3	*Ambrosia dumosa*	-	-	2.2	2.0	-	-
		Baileya multiradiata	-	-	-	11.4	-	26.7
		Ephedra nevadensis	-	-	-	-	2.6	-
		Eriogonum inflatum	-	-	8.7	-	-	-
		Erioneuron pulchellum	-	-	-	-	-	0.6
		Sphaeralcea ambigua	-	100.0	87.0	75.6	97.4	72.8
		Yucca schidigera	-	-	2.2	11.0	-	-
	CS4	*Ambrosia dumosa*	80.4	-	2.1	-	-	-
		Aristida purpurea	-	-	-	-	2.9	-
		Baileya multiradiata	8.7	-	9.6	24.5	33.7	3.3
		Ephedra nevadensis	-	-	27.2	-	0.5	-
		Eriogonum inflatum	-	-	8.6	-	-	0.4
		Eriogonum species	-	-	-	-	-	3.3
		Lycium species	-	-	2.4	-	-	-
		Opuntia basilaris	-	100.0	-	6.1	-	7.9
		Psorothamnus fremontii	-	-	22.8	-	-	-
		Sphaeralcea ambigua	8.7	-	27.3	55.6	62.9	85.1
		Stephanomeria pauciflora	-	-	-	13.8	-	-
		Yucca schidigera	2.2	-	-	-	-	-
	CS5	*Ambrosia dumosa*	-	-	9.2	2.7	2.8	-
		Baileya multiradiata	12.8	-	4.0	16.2	8.2	16.8
		Ephedra nevadensis	-	48.8	-	-	-	-

		Eriogonum inflatum	-	-	10.4	-	15.5	-
		Eriogonum species	-	-	-	-	-	8.9
		Erioneuron pulchellum	-	-	-	-	-	1.3
		Hymenoclea salsola	-	-	-	-	1.3	-
		Lycium species	-	-	-	-	1.3	-
		Machaeranthera pinnatifida	-	-	3.6	-	2.1	-
		Prunus fasciculata	-	-	-	-	3.8	-
		Psorothamnus fremontii	-	-	2.1	-	-	-
		Sphaeralcea ambigua	87.2	15.1	70.6	75.2	62.3	70.7
		Stephanomeria pauciflora	-	-	-	-	1.6	-
		Yucca schidigera	-	36.0	-	5.9	1.2	2.3
	CS6	*Ambrosia dumosa*	-	-	0.3	-	8.5	4.3
		Baileya multiradiata	-	-	-	2.8	-	19.9
		Eriogonum inflatum	-	-	-	23.2	-	13.3
		Krameria species	-	-	-	4.7	-	-
		Larrea tridentata	-	-	-	-	-	8.4
		Opuntia basilaris	-	-	10.9	-	3.3	-
		Sphaeralcea ambigua	100.0	100.0	87.2	69.2	87.2	54.1
		Yucca schidigera	-	-	1.6	-	1.0	-
	CS7	*Ambrosia dumosa*	19.8	100.0	16.9	7.5	78.8	30.4
		Larrea tridentata	79.3	-	9.7	14.3	4.2	-
		Lycium species	-	-	66.5	4.8	-	-
		Mirabilis multiflora	-	-	-	10.9	-	-
		Psorothamnus fremontii	-	-	-	60.5	-	-
		Sphaeralcea ambigua	0.9	-	7.0	2.0	17.0	69.6
Dry Rock	CS8	*Ambrosia dumosa*	68.3	-	23.3	-	6.8	-
		Baileya multiradiata	-	-	-	-	-	3.8
		Encelia species	-	-	6.4	-	12.7	-
		Ephedra nevadensis	-	-	1.7	-	-	-
		Eriogonum inflatum	-	-	3.1	-	7.8	-
		Gutierrezia species	-	-	3.1	-	4.1	-
		Hymenoclea salsola	7.4	-	2.6	-	5.5	-
		Krameria species	21.7	-	-	-	-	-
		Machaeranthera pinnatifida	-	-	-	-	1.6	-
		Mirabilis multiflora	-	-	4.2	-	-	-
		Psorothamnus fremontii	-	-	54.5	-	61.5	-
		Sphaeralcea ambigua	-	100.0	1.1	100.0	-	96.2
		Yucca schidigera	2.6	-	-	-	-	-
	CS9	*Ambrosia dumosa*	16.7	71.4	21.0	27.0	11.1	66.3
		Baileya multiradiata	8.5	-	4.3	-	4.2	-
		Coleogyne ramosissima	8.7	-	10.3	-	12.7	-
		Ephedra nevadensis	-	-	3.8	17.2	-	-
		Eriogonum fasciculatum	-	-	10.7	-	-	-
		Lycium species	-	-	0.2	-	1.5	5.2
		Prunus fasciculata	3.3	-	6.9	-	12.2	-
		Psorothamnus fremontii	33.4	-	9.4	25.2	36.6	4.8
		Sphaeralcea ambigua	-	28.6	5.8	30.6	7.2	23.7
		Stephanomeria pauciflora	-	-	1.8	-	-	-
		Yucca schidigera	29.3	-	25.7	-	14.5	-
	CS10	*Ambrosia dumosa*	50.8	-	6.7	47.1	7.3	40.5

		Baileya multiradiata	-	-	-	1.0	-	5.8
		Ephedra nevadensis	-	-	21.6	9.8	-	-
		Larrea tridentata	-	-	25.9	-	32.6	-
		Lycium species	-	-	3.9	13.7	8.7	-
		Mirabilis multiflora	-	-	1.4	-	-	-
		Psorothamnus fremontii	13.6	-	29.2	-	51.5	-
		Shrub	-	-	-	-	-	0.8
		Sphaeralcea ambigua	0.2	-	-	28.4	-	17.4
		Stephanomeria pauciflora	-	-	11.3	-	-	-
		Yucca schidigera	35.4	-	-	-	-	35.5
Garnet	CS2	*Ambrosia dumosa*	2.1	-	-	-	2.8	-
		Ephedra nevadensis	-	-	9.1	-	-	-
		Krameria species	28.6	-	50.5	4.3	16.1	-
		Larrea tridentata	69.3	-	40.4	71.7	81.1	43.1
		Pleuraphis rigida	-	-	-	12.3	-	10.3
		Sphaeralcea ambigua	-	100.0	-	11.6	-	46.6
Halfway	H1	*Acamptopappus sphaerocephalus*	-	-	-	2.1	-	1.8
		Ambrosia dumosa	14.0	13.5	10.2	14.1	22.1	10.1
		Aristida purpurea	-	-	-	-	-	1.9
		Baileya multiradiata	-	2.4	-	2.1	-	3.7
		Opuntia species	-	4.2	-	4.5	-	0.3
		Ephedra nevadensis	5.5	-	2.4	-	4.7	-
		Eriogonum fasciculatum	-	5.0	-	3.3	-	3.9
		Eriogonum inflatum	-	-	-	-	-	2.1
		Erioneuron pulchellum	-	-	-	2.6	-	2.4
		Gutierrezia species	-	-	1.8	4.3	-	4.4
		Hymenoclea salsola	-	-	-	-	-	4.5
		Krameria species	6.7	7.9	3.6	6.2	7.4	9.6
		Krascheninnikovia lanata	-	-	-	0.2	-	-
		Larrea tridentata	35.6	44.0	33.9	20.3	39.7	31.8
		Lycium species	-	-	-	0.4	-	-
		Prunus fasciculata	-	-	-	13.5	-	2.4
		Salvia dorrii	-	4.6	-	-	-	-
		Salazaria mexicana	-	-	-	-	-	0.3
		Shrub	-	2.0	-	-	-	-
		Sphaeralcea ambigua	20.8	4.6	30.5	8.3	26.2	9.3
		Stephanomeria pauciflora	-	2.5	4.9	10.8	-	3.2
		Thamnosma montana	-	3.5	-	5.8	-	8.4
		Yucca brevifolia	17.4	5.7	12.6	1.5	-	-
	H2	*Ambrosia dumosa*	-	27.4	13.1	35.2	1.9	14.2
		Baileya multiradiata	-	-	-	0.5	-	-
		Coleogyne ramosissima	-	-	-	15.9	-	-
		Ferocactus cylindraceus	-	-	-	-	7.2	-
		Larrea tridentata	23.5	-	58.6	5.5	63.7	30.8
		Sphaeralcea ambigua	76.5	-	28.3	6.0	24.8	15.7
		Stephanomeria pauciflora	-	-	-	-	2.3	-
		Yucca brevifolia	-	72.6	-	36.8	-	39.3
Meadow Valley	MV1	*Acamptopappus sphaerocephalus*	0.3	-	0.1	-	1.5	-
		Ambrosia dumosa	6.3	2.2	7.8	2.0	5.1	2.9
		Aristida purpurea	-	-	0.3	-	1.0	-

		Baileya multiradiata	-	-	1.7	1.0	-	1.1
		Coleogyne ramosissima	-	25.2	-	26.2	-	10.6
		Encelia species	-	-	-	0.4	-	1.4
		Ephedra nevadensis	-	1.5	-	1.9	-	1.3
		Eriogonum inflatum	-	-	11.5	1.6	25.6	3.1
		Eriogonum species	-	-	-	-	0.0	-
		Erioneuron pulchellum	1.3	-	1.6	-	1.7	0.3
		Ferocactus cylindraceus	-	1.3	-	0.8	-	1.2
		Gaura coccinea	-	-	-	0.0	-	1.0
		Krameria species	24.0	27.3	12.2	15.4	12.4	16.1
		Larrea tridentata	14.5	13.9	8.0	5.4	13.1	8.8
		Menodora spinescens	20.6	2.5	5.0	4.9	6.9	3.8
		Opuntia species	0.4	-	0.8	-	-	-
		Pleuraphis rigida	6.8	-	21.1	-	2.3	0.2
		Prunus fasciculata	-	5.9	-	7.5	-	14.1
		Psorothamnus fremontii	18.0	5.0	14.6	15.0	12.3	10.5
		Shrub	-	-	0.9	-	-	-
		Sphaeralcea ambigua	0.3	-	6.6	3.7	14.8	8.4
		Stephanomeria pauciflora	7.6	-	7.7	-	3.3	0.2
		Yucca schidigera	-	15.1	-	14.3	-	14.9
	MV2	*Ambrosia dumosa*	*	-	-	4.5	0.8	-
		Aristida purpurea	*	-	-	4.5	0.0	-
		Baileya multiradiata	*	-	2.2	0.4	0.9	3.2
		Coleogyne ramosissima	*	-	12.3	1.9	9.7	-
		Encelia species	*	-	0.3	-	-	-
		Ephedra nevadensis	*	-	2.8	-	4.9	2.6
		Eriogonum fasciculatum	*	-	1.9	0.4	2.0	-
		Eriogonum inflatum	*	-	17.9	-	17.1	29.5
		Eriogonum species	*	-	-	3.8	-	0.2
		Erioneuron pulchellum	*	2.2	0.0	-	1.9	1.1
		Gaura coccinea	*	1.9	0.5	0.6	0.9	0.2
		Krameria species	*	13.1	15.4	2.9	9.8	0.7
		Larrea tridentata	*	34.9	-	1.2	-	-
		Menodora spinescens	*	-	5.8	5.8	4.4	-
		Pleuraphis rigida	*	-	0.8	-	1.6	8.7
		Prunus fasciculata	*	-	16.0	2.5	15.5	-
		Psorothamnus fremontii	*	34.4	9.0	-	8.9	11.0
		Sphaeralcea ambigua	*	13.4	10.6	3.0	17.4	39.0
		Yucca schidigera	*	0.2	4.3	20.1	4.3	3.8
Tramp	T1	*Ambrosia dumosa*	-	10.4	70.8	5.4	31.1	11.5
		Ephedra nevadensis	-	2.5	-	8.8	-	6.6
		Krameria species	-	0.9	-	-	-	2.5
		Larrea tridentata	-	86.2	29.2	47.3	68.9	79.4
	T2	*Ambrosia dumosa*	-	-	-	-	-	16.1
		Eriogonum inflatum	-	-	100.0	-	-	-
		Larrea tridentata	-	100.0	-	100.0	64.2	83.9
		Sphaeralcea ambigua	-	-	-	-	35.8	-

Chapter 4: Delineation of Final Aerial Seeding Polygons

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INTRODUCTION

In the Emergency Stabilization (ES) Plan, the Burned Area Emergency Response (BAER) team called for aerial seeding a total of 47,000 acres on the Ely District Fires. They recommended seeding 10,000 acres with a Mesic Blackbrush Seed Mix, 26,200 acres with a Non-Wilderness Pinyon-Juniper (PJ) Seed Mix, and 10,000 acres with a Wilderness PJ Seed Mix (USDI National Interagency BAER Team 2005) (Appendix C). The BAER team delineated polygons for potential seeding (Figure 4-1). These polygons were sizeable, covering more than 82,000 acres (32,200 ha) of pinyon-juniper vegetation types and close to 50,000 acres (20,000 ha) of the mesic blackbrush vegetation type. They also included large unburned islands and a variety of post-fire responses. To meet the 47,000-acre seeding prescription, field staff from the Ely Bureau of Land Management (BLM) and the Eastern Nevada Landscape Coalition (ENLC) delineated smaller polygons within each vegetation type. In total, we delineated 30 seeding polygons across all four fires. Thirteen of these were seeded with the Non-Wilderness PJ Seed Mix, eight with the Wilderness PJ Seed Mix, and nine with the Mesic Blackbrush Seed Mix. Seeding polygons ranged from 103 acres to 5,589 acres in size. Less than 8% of the burned area was seeded. They prioritized areas for seeding using a variety of GIS and remote sensing-derived datasets, as well as on-the-ground reconnaissance conducted in August, November and December 2005.

Delineating Vegetation Communities for Aerial Seeding

The first step in delineating seeding polygons involved identifying areas that were formerly pinyon-juniper woodlands (for the two PJ seed mixes) and areas that were formerly mesic blackbrush communities. We defined burned pinyon-juniper woodlands as areas that generally had more than 10% tree cover pre-fire. ENLC staff used a variety of GIS layers to decipher burned locations containing trees, including: ReGap satellite-derived vegetation (classed as Great Basin Pinyon Juniper Woodlands); digital raster graphics (DRGs, 1:24,000 topographic quads—using green color for an indication of woodlands); and digital orthophoto quarter quadrangles (DOQQs, aerial photography). DOQQs proved to be the best GIS layer for delineating burned pinyon-juniper woodlands since based on ground-truthing they were the most accurate in delineating the presence of trees. One difficulty in this process was deciphering interior chaparral communities from pinyon-juniper woodlands. These two vegetation types often form a mosaic in the higher elevation portions of the SNC, and even dense pinyon-juniper stands can often contain an interior chaparral component of fire-adapted resprouting shrub species. We wanted to prioritize pinyon-juniper woodlands that had less of an interior chaparral component for seeding. The most suitable pinyon-juniper woodland sites were identified regardless of wilderness designation. Where sites occurred in designated wilderness, we used the Wilderness PJ Seed Mix. This seed mix was also applied to non-wilderness areas bordering wilderness.

Burned mesic blackbrush communities were more difficult to delineate in the office using GIS. It is hard to decipher different types of desert scrub communities using DOQQs. Furthermore, Regap does not specifically identify blackbrush as a vegetation type. Instead, it delineates a vegetation type called “Mojave Mid-Elevation Mixed Desert Scrub,” which includes blackbrush-dominated areas as well as other shrub species found at mid-elevations in the Mojave. We wanted to seed burned blackbrush communities rather than a different vegetation community that might be more fire-resilient, e.g. *Purshia glandulosa*-dominated shrublands. Ground reconnaissance was therefore crucial in differentiating burned blackbrush from other burned vegetation types. We decided to seed mesic rather than thermic blackbrush communities; treatments were more likely to succeed in mesic sites because they generally receive more moisture than thermic sites. Spatial soil mapping unit data were used to draw a boundary between mesic and thermic soil types.

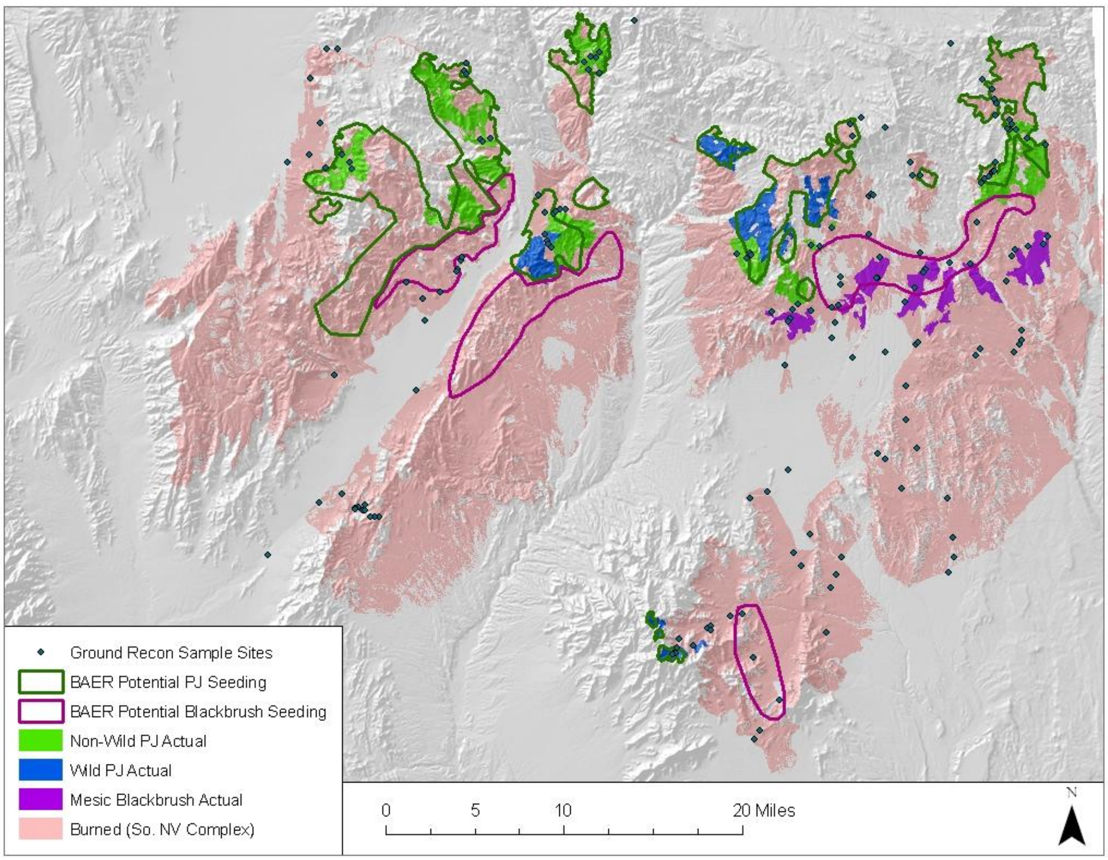

Figure 4-1. Comparison of BAER team potential seeding areas to polygons actually seeded. Reconnaissance photo points were used to delineate actual seeding polygons from the BAER team potential seeding areas.

Prioritizing Areas for Seeding

Within the burned pinyon-juniper woodlands and burned mesic blackbrush shrublands, we needed to identify the most appropriate areas for aerial seeding treatments. We wanted to concentrate on areas least likely to recover naturally (least likely to have naturally regenerating and/or surviving plant cover post-fire) and also had the highest chance of seeded species establishment. With these goals in mind, we decided to focus our seeding efforts on burned areas that had high fire-induced vegetation mortality and little post-fire green-up.

Although the BAER team provided many photos of the burned areas, these photos were not spatially referenced with coordinates and did not contain accompanying field notes. As a result, ENLC and Ely BLM staff took spatially referenced photographs and notes on pre-fire vegetation, soils, resprouting vegetation, and suitability for seeding at 168 sample points across the Ely District SNC fires (Figure 4-1). ENLC staff built a GIS database of photo points containing this information. The information was also used to assess the adequacy of available GIS layers for delineating seeding polygons.

The United States Geological Survey Earth Resources Observation and Science Center (USGS EROS) provided some additional GIS data layers that were essential in delineating seeding polygons. The first dataset was a burned/unburned layer that contained unburned islands greater than 30 m in diameter. We used this GIS layer to avoid placing seeding polygons in areas that had substantial unburned islands.

USGS-EROS also provided us with a variety of Landsat-derived NDVI (Normalized Difference Vegetation Index) and dNDVI (differenced Normalized Difference Vegetation Index) spatial datasets to extend our ground reconnaissance evaluations of post-fire green-up (e.g. Figure 4-2). These GIS layers, used in combination, proved very useful for locating areas that had high fire-induced vegetation mortality and little post-fire green-up.

NDVI is considered a good estimate of greenness or vegetative biomass. Higher values of NDVI should mean more vegetation. dNDVI is a measure of change in NDVI (greenness) from one time to another. In terms of post-fire vegetation response, NDVI should be an estimate of the total amount of vegetation on the site, whereas dNDVI should be an estimate of the amount of post-fire vegetation establishment. NDVI is calculated as follows: (NIR-R)/(NIR+R) where NIR (near infrared) and R (red) represent spectral reflectance in those regions of the electromagnetic spectrum.

We began by using dNDVI to display the change in greenness from immediately following the fire in July 2005 to September 2005. This time period was selected because the data was available and because we needed to identify seeding polygons prior to the winter application of seed. Based on ground reconnaissance, dNDVI proved useful for identifying

areas that had both high vegetation loss immediately after the fire and subsequent initial recovery of greenness. These areas generally had high dNDVI. They tended to include interior chaparral-dominated sites that burned rather completely but contained fire-adapted shrubs that began to resprout soon after fire. We therefore avoided seeding high dNDVI areas. Areas with low dNDVI values indicated little green-up, but dNDVI alone could not separate areas that had high vegetation mortality and little green-up from areas that had low vegetation mortality and little green-up.

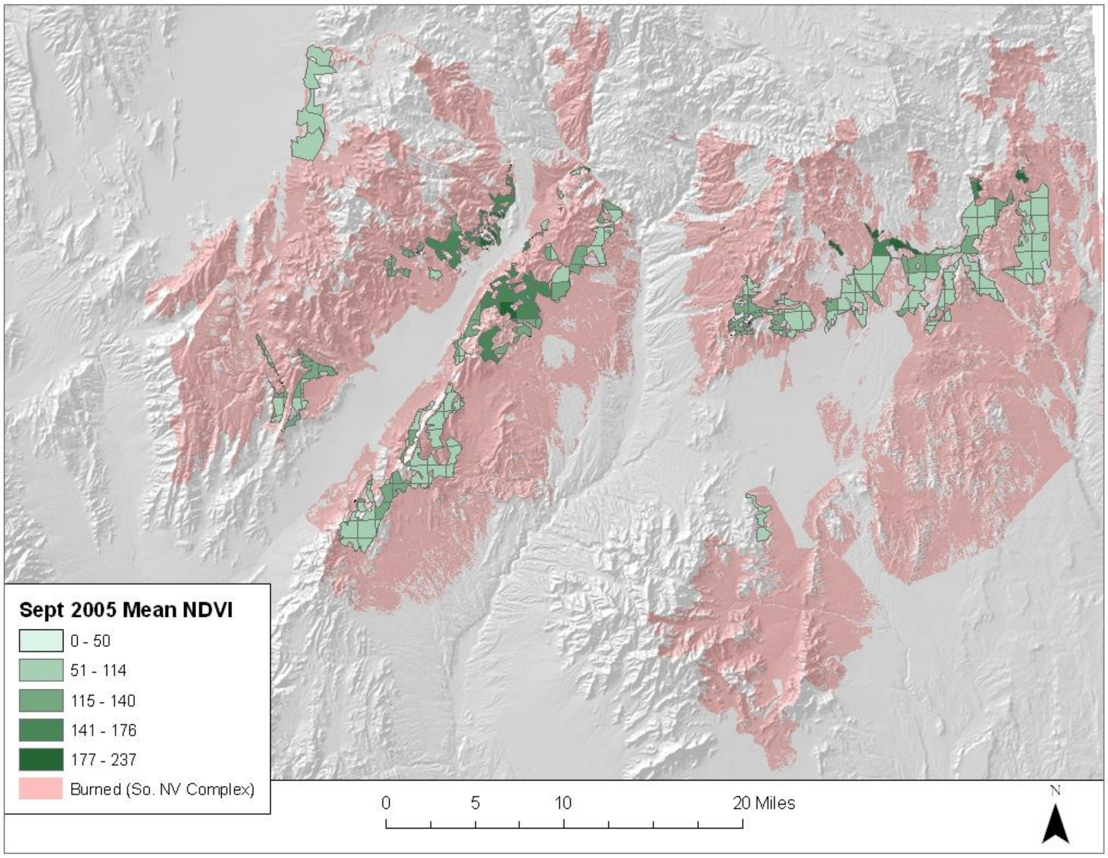

Figure 4-2. September 2005 mean NDVI within the Mesic Blackbrush Zone of the SNC.

We then used NDVI data from September 2005 to determine overall greenness for that time. This time period was selected because it allowed several months of potential green-up time while still being before our winter 2005/2006 seeding deadline. Areas with high NDVI values indicated high greenness. This could either be from low fire-induced vegetation mortality or significant post-fire green-up. We were not interested in seeding either of these types of areas. We therefore avoided seeding high NDVI areas. Areas with low NDVI values indicated low overall greenness, but NDVI alone could not separate areas with low greenness and little green-up from areas with low greenness, but some significant recent green-up.

We placed our seeding polygons in areas that had both high fire-induced vegetation morality and little post-fire green-up. Such areas would have a low NDVI value (indicating high vegetation mortality) and a low dNDVI value (indicating little post-fire green-up). Just prioritizing all areas with low NDVI values was not specific enough because it would have included areas that showed a subsequent green-up post-fire. Just prioritizing all areas with low dNDVI was not specific enough because it would have included areas that had low vegetation mortality. By using NDVI and dNDVI together, we were able to target only areas with high vegetation morality and little post-fire green-up (Table 4-1).

Table 4-1. Prioritization of seeding polygons using NDVI and dNDVI. Areas with low September NDVI and low dNDVI were the highest priority for seeding.

		September NDVI	
		Low	**High**
dNDVI (Sept-July)	**Low**	Areas with high vegetation mortality and little post-fire green up. HIGH priority for seeding.	Areas with low vegetation mortality and little post-fire green-up. LOW priority for seeding.
	High	Areas with high vegetation mortality and significant post-fire green up. LOW priority for seeding.	Areas with low vegetation mortality and/or high post-fire green up. LOW priority for seeding .

The process for identifying seeding polygons was not perfect. The GIS data used was coarse in scale. While we conducted rather extensive ground reconnaissance of the burned areas, individual areas proposed for treatment could not be thoroughly evaluated. As a result, a small portion of the mesic blackbrush seeding treatment dipped into potential desert tortoise habitat.

SAMPLING DESIGN FOR AERIAL SEEDING AND NATURAL REGENERATION TREATMENT EFFECTIVENESS MONITORING

The following description of field sampling design for aerial seeding and natural regeneration treatments can be used for understanding the sampling methods behind the analyses presented in Chapters 5-9.

Following the application of the aerial seeding treatments, the USGS Western Ecological Research Center (WERC), BLM, and ENLC drafted and agreed upon a sampling design for the SNC aerial seeding and natural regeneration treatments. We reached this design through compromise and designed three primary sampling methodologies that would complement each other. One methodology was more intensive and designed to specifically compare seeded and unseeded conditions in demonstration plots. The second methodology

was more extensive and designed to maximize sampling coverage of the aerial seeding polygons. The third methodology was used to evaluate the majority of the burned area that was not seeded, i.e. natural regeneration. Due to time and funding constraints, some of the sampling methodologies had to be modified over the three-year sampling period. While changes needed to be made to accommodate real funding and time issues, these changes may have weakened the strength of some of our analyses. In addition to the three primary sampling methodologies, we also conducted qualitative assessment write-ups and ocular cover estimates (Remote Sensing plots) in 2008 to help ground truth the remote sensing portion of this project.

Demonstration Plots: Comparing Seeded and Unseeded Conditions

We used paired 40-acre areas called Demonstration (DM) plots to evaluate the effectiveness of the aerial seeding treatments. Each pair consisted of one seeded DM plot and one unseeded control DM plot. We placed DM plot pairs in generally adjacent areas with similar environmental conditions including soil type, pre-fire vegetation, slope, aspect, distance from roads, and when possible burn severity. We created one pair of DM plots for 28 of the 30 seeding polygons. The exceptions were two very small seeding polygons on the Halfway Fire Wilderness PJ Seeding Treatment. For large seeding polygons, we created additional pairs of DM plots at a rate of approximately one per 1000 acres of aerial seeding.

Within each DM plot, we created five replicate brushbelt (BB) macroplots (see below for description). These BB macroplots were randomly located within the DM plots, and inside a 50-m buffer from the edge of the unseeded control DM plot in order to minimize the possibility of seed drift into control plots (Figure 4-3). BB macroplots within unseeded control DM plots were located 50-200 m from the edge of the seeding treatment. It is likely that some seed drift into control plots still occurred. We created a total of 76 DM plots and 380 BB macroplots within them. We sampled 92% of DM plots in 2006 and 100% of DM plots in 2007 and 2008. We sampled 66-70% of the BB macroplots each year (Table 4-2). In 2006, we attempted to sample all of the BB macroplots in every DM plot, but we ran into time and funding constraints. In years 2007 and 2008, crews sampled only three to four replicate BB macroplots in each DM plot to ensure that all DM plots would have some sampled BB macroplots.

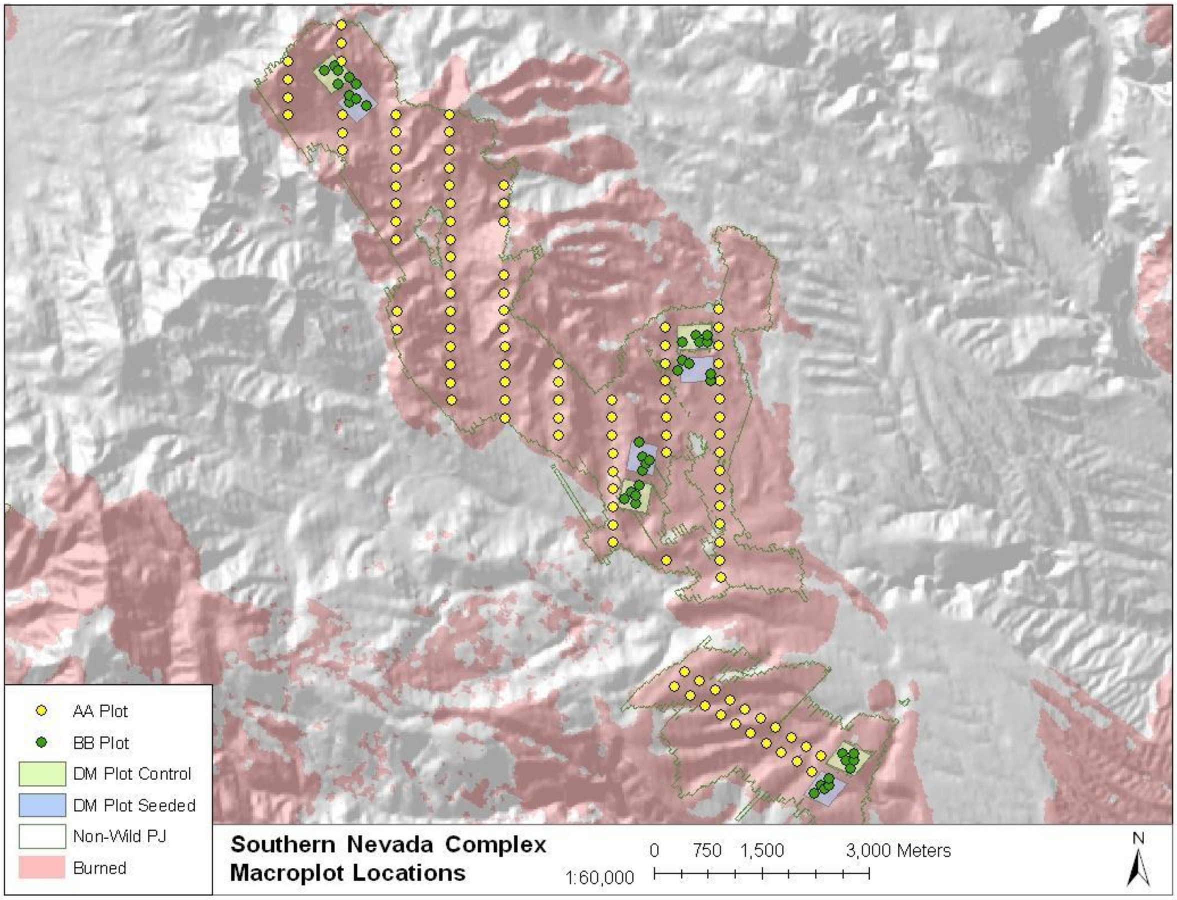

Figure 4-3. An example of the placement of macroplots within a seeding polygon. Brushbelt (BB) macroplots are located within paired demonstration (DM) plots. Additional Aerial Seeding Coverage (AA) macroplots are placed in transects throughout the seeding polygon.

Table 4-2. Total number of plots completed in the SNC fires broken down by treatment type, plot type and year. Treatment types are: NR = natural regeneration; MBB = mesic blackbrush; NWPJ = non-wilderness pinyon-juniper; WPJ = wilderness pinyon-juniper. Plot types are: DM = demonstration; BB = brushbelt; AA = additional aerial seeding coverage.

	NR	**MBB**		**NWPJ**		**WPJ**		**Totals**
		Seeded	***Control***	***Seeded***	***Control***	***Seeded***	***Control***	
All Plots Created								
DM Plots	0	10	10	20	20	8	8	76
BB Macroplots	58	50	50	100	100	40	40	438
AA Macroplots	0	249	0	510	0	232	0	991
2006 Plots Sampled								
DM Plots	0	9	9	19	19	7	7	70
BB Macroplots	39	40	39	69	68	25	25	305
AA Macroplots	0	129	0	191	0	68	0	388
2007 Plots Sampled								
DM Plots	0	10	10	20	20	8	8	76
BB Macroplots	58	31	30	65	63	24	24	295
AA Macroplots	0	26	0	285	0	75	0	386
2008 Plots Sampled								
DM Plots	0	10	10	20	20	8	8	76
BB Macroplots	58	30	30	62	61	24	24	289
AA Macroplots	0	109	0	394	0	156	0	659

Brushbelt (BB) Macroplots. The brushbelt (BB) macroplot was adapted from the Fire Monitoring Handbook (USDI National Park Service 2003). The BB macroplot is a permanently monumented 5 x 30 m rectangle in which field crews collected data on herbaceous density, shrub density, ocular cover/additional species, point-intercept cover, basal gap and composite burn index (Figure 4-4). The two 30-m sides of the brushbelt are termed the primary and secondary transects, respectively. The ends of the primary transect are termed 0P (starting point at 0 m on the primary transect) and 30P (ending point at 30 m on the primary transect). Each end of the primary transect is monumented with rebar. The starting point (0P) is also marked with a yellow fiberglass fencepost to aid in locating the brushbelt. Crews photographed the primary transect of the brushbelt at 0P and at 30P each year. Crews photographed each density frame in 2007 and 2008.

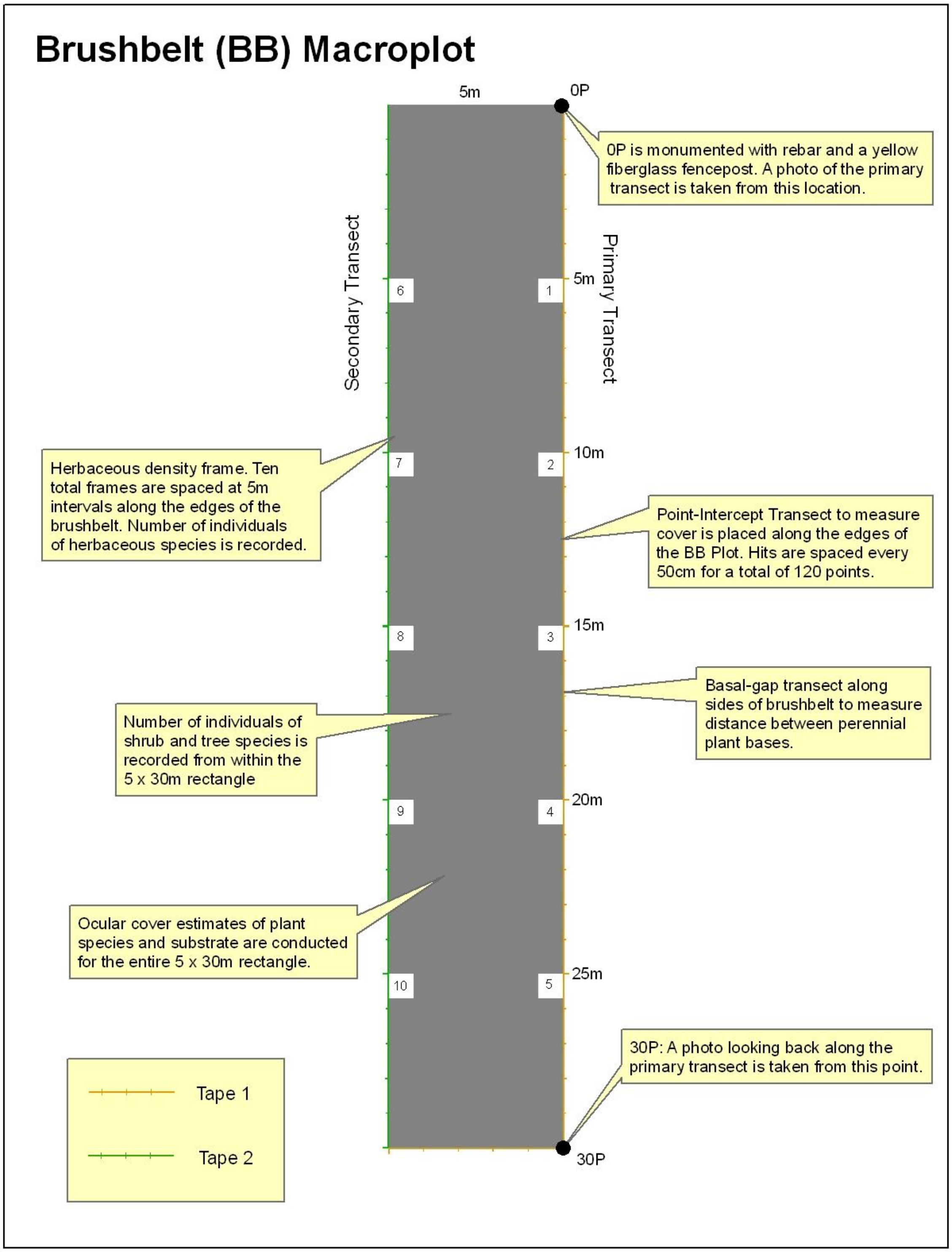

Figure 4-4. Overview of a brushbelt (BB) macroplot. The 5 x 30 m BB macroplot includes subsamples for herbaceous density, shrub density, ocular cover/additional species, point-intercept cover, basal gap and composite burn index.

Herbaceous Density. Field crews placed ten 0.5-m^2 quadrats at 5 m intervals along the primary and secondary transects. Crews recorded the number of individuals of all herbaceous species and shrub seedlings living during the current growing season. Crews counted annual grasses by culm due to the difficulty determining what constitutes a single individual. Crews also differentiated perennial plants by age class. Age classes include: 1) immature-seedling, 2) mature, 3) resprout, and 4) top-kill (Table 4-3). For each group (species-age class), crews recorded an average height in meters.

Table 4-3. Age descriptions used on the SNC fires. Each perennial plant recorded in herbaceous density and shrub density was given an age class.

Code	Definition
Immature-seedling	A young, immature plant that is not firmly rooted in the ground. Its root system is not well established and can be pulled out of the soil relatively easily, by the mouth of a cow for instance. Pulling on the plant with your hand is a good test for this.
Mature	A plant that is firmly rooted in the ground and has stabilized the surrounding soil from erosion. The plant cannot easily be pulled up by the roots.
Resprout	A plant that experienced aboveground mortality, but survived underground and is resprouting. In the second and third years post-fire, it may be difficult to tell if a plant is resprouting. If you can't tell, just record it as "mature."
Top Kill	A plant that has experienced herbivory-induced aboveground mortality post-fire.

Shrub Density. Field crews recorded the number of individuals for all mature shrub and tree species from within the entire 5 x 30 m brushbelt rectangle. They grouped individuals into age classes (Table 4-3). Numbers of individuals of shrub and tree seedlings were not recorded since these were recorded in the herbaceous density subsample. Dead (unburned) shrubs and burned shrubs within this subsample were also recorded. If crews could not identify the species of a dead or burned shrub, they lumped them as either dead shrub or burned shrub respectively.

Ocular Cover/Additional Species. Crews recorded ocular cover estimates for species (living and dead), litter, and substrate (Table 4-4). They estimated cover in 5% intervals, but if two or more species added up to 5% then they split these species into smaller increments. The point-intercept method (see below) was used to help standardize their ocular estimates approximately once a week. Field crews also recorded any additional species found within the brushbelt but not found in the other subsamples as a part of this protocol. This became the primary cover protocol for the 2007 and 2008 growing seasons. We made the switch from point-intercept to ocular cover estimation following the 2006 growing season due to funding and time constraints.

Table 4-4. Substrate and non-plant species codes. These codes were used in ocular cover and point-intercept cover subbelts to assess non-live plant cover.

Code	Description
Substrate Codes	
BARE	Bare soil
DUFF	Duff
RFRAG	Rock fragments less than 5cm in diameter
ROCK	Rocks larger than 5cm in diameter
WATER	Water
Non-plant Codes	
BIOCRUST	Biological soil crusts
BURNSHB	Unidentifiable dead standing burned shrubs
DEADFRB	Unidentifiable dead standing forbs
DEADGRS	Unidentifiable dead standing grass
DEADSHB	Unidentifiable dead standing unburned shrubs
DEADTRE	Unidentifiable dead trees
FUNGUS	Fungus
LICHEN	Lichen
LITTER	Detached dead plant parts
MOSS	Moss
NOVEG	No vegetation present
SCAT	Scat
STUMP	Stump
WDGT25	Woody litter, diameter larger than 25cm
WDLT25	Woody litter, diameter smaller than 25cm

Point-Intercept Cover. Field crews conducted point-intercept cover transects to estimate plant species and substrate cover. The transects were placed along the primary and secondary transects of the BB macroplots. Readings occurred every 50 cm, for a total of 120 points along 60 m of transect. Crews took readings using optical point projection devices. They conducted point-intercept cover on all sampled brushbelts during the 2006 season and on a majority of brushbelts within the wilderness and non-wilderness pinyon-juniper treatments in 2007. These 2007 samples have both point-intercept and ocular cover estimates.

Basal Gap Transect. Crews measured the distance in centimeters between perennial plant bases (both alive and dead) along the primary and secondary transects of the brushbelts using a line-intercept technique. They only included perennial plant bases that directly intercepted the transect. Gaps less than 20 cm in length were not recorded.

Composite Burn Index (CBI). In 2006, crews estimated ecological burn severity using the CBI technique (Key and Benson 2003). This method utilizes a 30-m diameter circle superimposed on top of the brushbelt rectangle. It evaluates changes from pre-burn to post-burn conditions in the first growing season post-fire at various strata including substrates, herbs, low shrubs and tall shrubs.

Additional Aerial Seeding Coverage (AA) Macroplots. We located transects of AA macroplots within seeding polygons to gain better coverage of the seeding treatments. We placed transects across seeding polygons to maximize coverage of environmental heterogeneity including topographical variables, soils, and pre-fire vegetation. We spaced AA macroplots at 250 m intervals along these transects. We created a total of 991 AA macroplots. We sampled 39-66% of the total AA plots each year (Table 4-2). Each AA macroplot consists of a 30-m diameter circle (Figure 4-5). The circle is bisected by two perpendicular transects that intersect at the circle's center. Field crews took a photo of the primary transect each year from the starting point (0P).

Density. In each AA macroplot, crews placed four 0.5-m^2 quadrats on the primary and secondary transects of the AA plot. In each quadrat, crews counted the number of individuals by species (including herbaceous and woody species). Crews differentiated perennial species by age class (Table 4-3). Crews photographed each density quadrat in 2007 and 2008.

Ocular Cover. Crews estimated percent cover of plant species and substrate with at least 5% cover within the 30-m diameter circle. If two or more plant species added up to 5% cover, crews used smaller increments for these species. Additionally, crews would make a quick, but not necessarily complete, list of additional species found within the circle. As a time consideration, unknown plants encountered outside of the density quadrats but within an AA plot were not collected for later identification or listed in additional species. Any seeded or invasive species were always to be included as additional species.

Composite Burn Index (CBI). In 2006, crews estimated ecological burn severity using the CBI technique (Key and Benson 2003) from within the 30-m diameter circle (see description above).

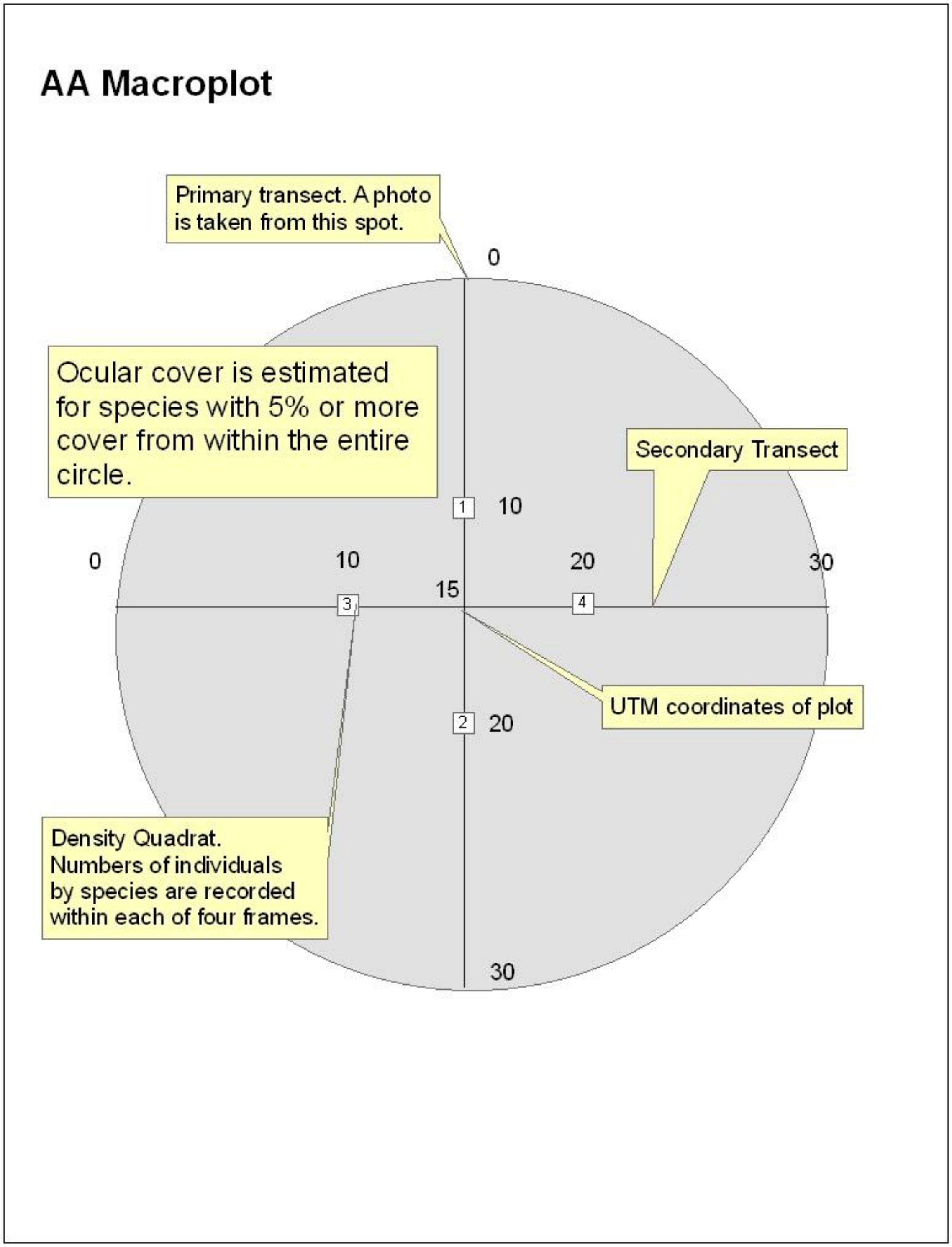

Figure 4-5. Overview of an AA macroplot. The 30 m radius AA macroplot includes subsamples for density, ocular cover and composite burn index.

Natural Regeneration Treatment Effectiveness Monitoring

Approximately 550,000 acres of the Ely District SNC fires were not aerially seeded. These areas are part of the natural regeneration treatment. We needed to evaluate these areas to determine if not seeding was an acceptable choice, and when normal management practices, such as livestock grazing, could be returned to the burned areas. We established a total of 58 BB macroplots within the natural regeneration areas (see above for description). We used a stratified random sampling design for these plots. We stratified by grazing allotment and pre-fire vegetation type. We randomly located BB macroplots only in creosote and thermic blackbrush vegetation types. We excluded areas that were not in these vegetation types, areas in steep mountainous terrain that cattle do not normally access, and areas more than one mile from roads.

Additional Monitoring Methodologies

Remote Sensing (RS) Plots. In 2008, we established and monitored an additional set of plots to try to increase our coverage of non-seeded burned areas to aid in the remote sensing analysis of this project. The RS plot consisted of ocular cover estimates from a 30-m circle and a photo. The plot is similar to an AA plot but without the density subsample. We monitored a total of 31 RS plots from the Meadow Valley, Delamar, and Duzak fires.

Qualitative Assessments (QAs). In 2008 we completed a total of 78 QAs from the SNC fires. The QA consists of a spatially referenced set of photos and qualitative write-up of post-fire conditions filled out on an assessment form. We included written descriptions of soil/topographic properties, burn severity, likely pre-fire vegetation, post-fire dominance, post-fire diversity, effectiveness of macroplots in capturing the common post-fire conditions, abundance of annual grasses, seeded species establishment, and other notable factors within the areas. We completed QAs in both seeding polygons and in unseeded natural regeneration areas. We attempted to complete at least one QA per seeding polygon and in some areas we have multiple QAs per seeding polygon.

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USDI National Interagency Burned Area Emergency Response Team. 2005. Southern Nevada Complex Burned Area Emergency Stabilization Plan. Mesquite, NV, USA. 212 p.

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Chapter 5: Establishment of Aerial Seeding Treatments in Blackbrush and Pinyon-Juniper Sites Following the 2005 Southern Nevada Complex

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INTRODUCTION

In this chapter we specifically evaluate the level of establishment by seeded species that were aerially seeded into mesic blackbrush, wilderness, and non-wilderness pinyon-juniper sites within the Delamar, Duzak, Halfway, and Meadow Valley fire perimeters of the 2005 Southern Nevada Complex. Our goals were to determine if: (1) the seeded species attained thresholds of successful seeding density of three individuals m^{-2} in mesic blackbrush communities and five individuals m^{-2} in pinyon-juniper communities over the three year duration of the study, (2) diversity, frequency, and density of the seeded species increased over the three year duration of the study, and (3) diversity, frequency, and density of the seeded species differed among vegetation types and between seeded and unseeded areas over the three year duration of the study. We then discuss the patterns and their implication for future post-fire seeding projects in similar ecotypes of the Mojave Desert. Because the seed mixes were associated with particular vegetation types we consider the terms seed mix and vegetation type to be synonymous in this chapter. Effects of hand-seeding treatments in the lower elevation creosotebush scrub zone are reported elsewhere in this report.

METHODS

Site Description and Study Design

Chapters 1 and 4 of this report contains details about the 2005 Southern Nevada Complex fires, the reasons that seeding treatments and other Emergency Stabilization and Rehabilitation (ES&R) actions were taken, the characteristics of the areas that were monitored, the post-fire seeding treatments that were implemented, the sampling design, and the sampling schedule. In the current chapter we only use data from the BB plots, which were sampled more intensely and included both seeded and unseeded control areas. Data from the AA plots, which were sampled less intensively and distributed more extensively, but did not include unseeded control areas, were only used as ancillary observations in this chapter. We also confined our analyses to species richness, frequency of occurrence, and density of the seeded species. Cover data were not included in the analyses because we did not expect plants to establish and produce measurable cover within the first 3 postfire years. For perennial species we only considered plants that were identified as seedlings, and did not include any plants identified as adults or resprouts because there was very little chance that plants could have originated from the seedings and grown to adults during the first 3 postfire years. We also only included data collected within replicate 0.5 m^2 herbaceous plots which were specifically designed to detect herbaceous species and seedlings of perennial species. We did not include data collected within the larger 5 x 30 m brushbelt transects which were designed to measure mature perennial plants. Our objective was to directly focus the analyses on comparisons of establishment by seeded species in seeded versus unseeded areas and evaluate how these comparisons varied among vegetation types.

Data Analysis

The Kolmogorov-Smirnov test (KS test) was used to test for differences in the distributions of total species number and total density per plot (species identity and stems per m^2 pooled across species, respectively) between seeded and unseeded conditions. The KS tests were done for all species pooled across vegetation types and separately for each vegetation type.

Multiway frequency tables (MFT's; Tabachnik and Fidell 1996) were used to analyze the frequency of species in the seeded/unseeded-vegetation type conditions. MFT's do not assume that data have a particular distribution or error structure (e.g. Gaussian for least-squares ANOVA or Poisson in GLM's) or that variances between different conditions are equal. Because a very high proportion of the plots in seeded and unseeded conditions had no seeded species, the data had non-normal distributions and dissimilar variances among vegetation types and seeded/unseeded conditions. MFT allowed the expected frequencies to be weighted by the proportional occurrence of plots in the different seeded/unseeded-vegetation type combinations. The analysis was constructed as a three-way frequency table with the simple and interactive effects of seeding (seeded, unseeded), vegetation type (mesic blackbrush, non-wilderness pinyon-juniper, wilderness pinyon-juniper) and postfire year (1, 2, 3), with counts of the number of species as the cells in the matrix.

Multi-level models were used to analyze differences over time among the seeding-vegetation type combinations for each of three response variables; absolute density of the seeded species ($\log_e$ stems per m^2), relative density of the seeded species (absolute density divided by the total stem density in a plot), and Simpson's index ($-\log_e D$). Multi-level models are a very flexible and robust set of methods based on maximum-likelihood estimates of parameters, rather than ordinary least square estimates such as used in rmANOVA (Gelman and Hill 2007). Because they are based on maximum likelihood estimates, multi-level models can handle situations where not all plots are sampled each year, which was the case in this project (conventional repeated-measures analysis of variance requires that all sample units in the analysis are sampled at each time occasion). Multi-level models also partition variance based on random and fixed factors. Fixed factors in the models included year and $year^2$ (the quadratic function of year) vegetation type (mesic blackbrush, non-wilderness pinyon-juniper, and wilderness pinyon-juniper), seeding (seeded or unseeded), and all two and three-way interactions. Fire (Delamar, Duzak, Halfway, and Meadow Valley) was considered a random factor. Akaike's Information Criteria (AIC) was used to determine the model with the greatest support from the pool of 14 possible models. Significance of parameters in the best supported model was tested with a Z-statistic (calculated by dividing the parameter estimate by its model-based standard error). The models used a Gaussian error structure and identity link for all three response variables. Absolute density was $\log_e$

transformed and relative density arcsin (i.e. angular) transformed to try to normalize residuals.

Simpson's index $-\log_e D$ takes into account both the number and relative abundance of species in a sample and has been found to be particularly useful for analyzing trends within species in a common group (Buckland et al 2005). Statistical analyses for the individual seeded species would have been extremely tenuous because they occurred at such low abundance and restricted distributions (see RESULTS). However, using Simpson's index allowed the data on the individual species to be synthesized into composite trends and patterns. *D* is the basic Simpson index:

$$D = \sum^{S}_{i=1} n_i(n_i-1)/N(N-1)$$

where S is the total number of species in a sample, n_i is the number of individuals of $species_i$, and N is the total number of individuals in the sample. Transforming the index by $-\log_e$ makes the index ≥ 0 and results in more diverse plots having higher values (more diverse plots have lower values of D if it is not transformed).

RESULTS

The KS tests indicated that there were no significant differences in the distributions of either species richness or density of seeded species between seeded and unseeded conditions for any of the vegetation types ($P \geq 0.213$). Across all vegetation types and years, seeded species occurred in 26% of the unseeded plots and 34% of the seeded plots. In the plots where the seeded species occurred, 74% had only one species present and 82% (seeded plots) to 97% (unseeded plots) had densities of < 1 stem m^{-2} (Figure 5-1). Four of the sixteen seeded species, *Atriplex canescens*, *Elymus wawawaiensis*, *Grayia spinosa*, and *Kochia prostrata*, were not recorded in any of the plots during any of the years (Table 5-1). Many plots that had seeded species present during the first postfire year did not have them present during subsequent years, and vice versa. Thus, there was no consistent trend over time in establishment and persistence of seeded species within the plots. We therefore present averages over 3 postfire years where appropriate in the summary statistics presented below.

Among the seeded species that were detected, *Elymus elymoides*, *Poa secunda*, and *Agropyron cristatum* were the most widespread occurring in 11.9%, 8.7%, and 6.7% of all plots averaged over all 3 years (Table 5-2). Each of these species along with *Pleuraphis jamesii* also showed increasing trends of plot occupancy over the 3 years, but the highest value was only 15.2% by year 3 for *Elymus elymoides*. These same four species were the only ones of the twelve seeded species that displayed increasing trends in stem density over the 3 years, but this peaked at only 0.045 stems/m^2 (approximately 1 stem per 22 square

meters) during year 3 for *Elymus elymoides*. Finally, seeded species comprised a very small percentage of the total stem density in the sampling plots, peaking at only 0.92% for *Agropyron cristatum* during year 3. Thus, across all plots and vegetation types, most sampling plots did not have seeded species present within them, their stem densities were exceedingly low, and the vast majority of the stems present in the plots were of species that were not included in the seed mixes.

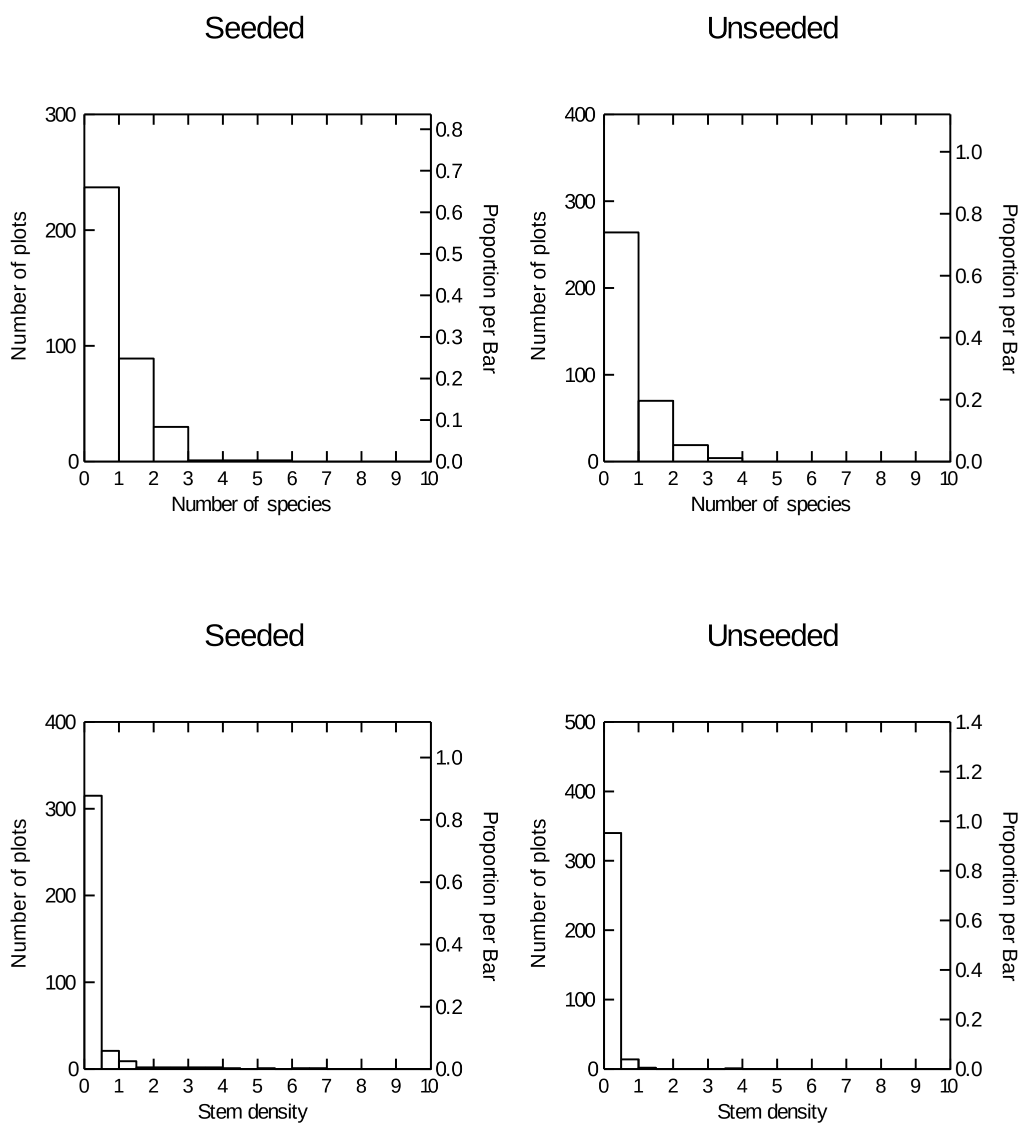

Figure 5-1. Histograms of the number of seeded species and their estimated stem density (stems per m^2) between seeded and unseeded conditions in the Southern Nevada Complex fires of 2005.

Table 5-1. Species seeded into three different vegetation types in the Southern Nevada Complex fires of 2005. MBB = mesic blackbrush, NWPJ = non-wilderness pinyon-juniper, WPJ = wilderness pinyon-juniper. A “1” indicates the species was seeded in that vegetation type. *Species that were seeded but did not appear in any of the BB monitoring plots and are, therefore, not included in subsequent tables.

Species	**Native**	**Life History**	**Life Form**	**MBB**	**NWPJ**	**WPJ**
Achnatherum hymenoides	Native	Perennial	Grass	1	1	1
Agropyron cristatum	Non-native	Perennial	Grass	0	1	0
Agropyron fragile	Non-native	Perennial	Grass	0	1	0
*Atriplex canescens**	Native	Perennial	Shrub	1	0	0
Elymus elymoides	Native	Perennial	Grass	1	1	1
Elymus lanceolatus	Native	Perennial	Grass	0	1	0
*Elymus wawawaiensis**	Native	Perennial	Grass	0	1	0
*Grayia spinosa**	Native	Perennial	Shrub	1	0	0
Hesperostipa comata	Native	Perennial	Grass	0	0	1
*Kochia prostrata**	Non-native	Perennial	Sub-shrub	1	0	0
Linum perenne	Native	Perennial	Forb	1	0	0
Penstemon palmeri	Native	Perennial	Forb	0	1	1
Pleuraphis jamesii	Native	Perennial	Grass	1	0	0
Poa secunda	Native	Perennial	Grass	1	1	1
Sanguisorba minor	Non-native	Perennial	Forb	1	0	0
Sporobolus cryptandrus	Native	Perennial	Grass	1	0	0
Total				10	8	5

Table 5-2. The number of plots, mean stem density, percentage of plots, and the percentage of stems for twelve species seeded into areas that burned in the Southern Nevada Complex fires of 2005.

Species		**Year**		
	2006	**2007**	**2008**	**Overall**
Plots				
Achnatherum hymenoides	17	3	6	26
Agropyron cristatum	1	16	31	48
Agropyron fragile	2	0	0	2
Elymus elymoides	18	33	34	85
Elymus lanceolatus	2	0	0	2
Hesperostipa comata	2	0	2	4
Linum perenne	0	1	0	1
Penstemon palmeri	13	4	10	27
Pleuraphis jamesii	0	6	6	12
Poa secunda	21	16	25	62
Sanguisorba minor	0	3	2	5
Sporobolus cryptandrus	0	0	7	7
Stem density (per m^2)				
Achnatherum hymenoides	0.053	0.001	0.004	0.021
Agropyron cristatum	0.002	0.099	0.188	0.092
Agropyron fragile	0.005	0.000	0.000	0.002
Elymus elymoides	0.017	0.028	0.045	0.029
Elymus lanceolatus	0.003	0.000	0.000	0.001
Hesperostipa comata	0.008	0.000	0.004	0.004
Linum perenne	0.000	0.005	0.000	0.002
Penstemon palmeri	0.029	0.009	0.015	0.018
Pleuraphis jamesii	0.000	0.017	0.034	0.016
Poa secunda	0.058	0.033	0.025	0.040
Sanguisorba minor	0.000	0.015	0.003	0.006
Sporobolus cryptandrus	0.000	0.000	0.033	0.010
Plots (%)				
Achnatherum hymenoides	6.6	1.3	2.7	3.6
Agropyron cristatum	0.4	6.9	13.8	6.7
Agropyron fragile	0.8	0.0	0.0	0.3
Elymus elymoides	6.9	14.2	15.2	11.9
Elymus lanceolatus	0.8	0.0	0.0	0.3
Hesperostipa comata	0.8	0.0	0.9	0.6
Linum perenne	0.0	0.4	0.0	0.1
Penstemon palmeri	5.0	1.7	4.5	3.8
Pleuraphis jamesii	0.0	2.6	2.7	1.7
Poa secunda	8.1	6.9	11.2	8.7
Sanguisorba minor	0.0	1.3	0.9	0.7
Sporobolus cryptandrus	0.0	0.0	3.1	1.0

Table 5-2 continued.

Species	Year 2006	2007	2008	Overall
Stem density (%)				
Achnatherum hymenoides	0.11	0.00	0.01	0.04
Agropyron cristatum	0.03	0.83	0.92	0.58
Agropyron fragile	0.10	0.00	0.00	0.04
Elymus elymoides	0.03	0.19	0.10	0.10
Elymus lanceolatus	0.06	0.00	0.00	0.02
Hesperostipa comata	0.00	0.00	0.00	0.00
Linum perenne	0.00	0.05	0.00	0.02
Penstemon palmeri	0.28	0.03	0.09	0.14
Pleuraphis jamesii	0.00	0.12	0.09	0.07
Poa secunda	0.44	0.07	0.07	0.20
Sanguisorba minor	0.01	0.00	0.00	0.00
Sporobolus cryptandrus	0.00	0.09	0.13	0.07

Most of the seeded species occurred in both unseeded and seeded plots and in vegetation types where they were not intentionally seeded (Tables 5-3 and 5-4). However, some species trended towards higher frequencies in seeded than unseeded areas within the vegetation type where they were included in the seed mix (Table 5-3). Notable examples for mesic blackbrush included *Achnatherum hymenoides* which appeared in more seeded (18.6%) than unseeded (9.5%) plots, *Sanguisorba minor* seeded (11.6%) unseeded (0%), and *Sporobolus cryptandrus* seeded (14.0%) unseeded (2.4%). For non-wilderness pinyon-juniper, *Agropyron cristatum* appeared in more seeded (36.8%) than unseeded (26.3%) plots, and for wilderness pinyon-juniper *Achnatherum hymenoides* appeared in more seeded (21.4%) than unseeded (7.1%) and *Pleuraphis jamesii* seeded (32.1%) and unseeded (10.7%).

Density of the seeded species was extremely low during all years, vegetation types, and fires (Tables 5-5 and 5-6). Between 2006 and 2008 only 37 plots (4.3%) had densities of seeded species that exceeded 1 m^{-2}. Despite the generally low density of seeded species, patches of relatively high density did occur in a small proportion of the seeded plots. For example, in the Meadow Valley fire there were several seeded plots in non-wilderness pinyon-juniper where seeded species comprised > 30% of the herbaceous density at levels of 7.6 individuals m^{-2}. However, it must be noted that patches of similarly high density of seeded species also occurred in unseeded plots within the same fire and vegetation type, making it impossible to attribute observations of high density patches of seeded species to the seeding treatment alone.

Table 5-3. The number (N) and percentage (%) of plots that twelve species occurred in for seeded and unseeded conditions in three vegetation types that burned in the Southern Nevada Complex fires of 2005. **Bold** numbers identify the vegetation types a species was seeded into. Vegetation types are: MBB = mesic blackbrush, NWPJ = non-wilderness pinyon-juniper, and WPJ = wilderness pinyon-juniper.

	MBB		NWPJ		WPJ		Total	
Species	N	%	N	%	N	%	N	%
Unseeded								
Achnatherum hymenoides	4	9.5	2	2.6	2	7.1	8	5.5
Agropyron cristatum		0.0	20	26.3		0.0	20	13.7
Agropyron fragile		0.0	1	1.3		0.0	1	0.7
Elymus elymoides		0.0	29	38.2	11	39.3	40	27.4
Elymus lanceolatus		0.0		0.0		0.0	2	1.4
Hesperostipa comata		0.0		0.0	2	7.1	2	1.4
Linum perenne		0.0		0.0		0.0	0	0
Penstemon palmeri		0.0	12	15.8		0.0	12	8.2
Pleuraphis jamesii		0.0		0.0	3	10.7	3	2.1
Poa secunda		0.0	28	36.8	5	17.9	33	22.6
Sanguisorba minor		0.0		0.0		0.0	0	0
Sporobolus cryptandrus	1	2.4		0.0		0.0	1	0.7
N (species)	2		6		5		10	
Seeded								
Achnatherum hymenoides	8	**18.6**	4	**5.3**	6	**21.4**	18	12.2
Agropyron cristatum		0.0	28	**36.8**		0.0	28	19
Agropyron fragile		0.0	1	**1.3**		0.0	1	0.7
Elymus elymoides		0.0	32	**42.1**	13	**46.4**	45	30.6
Elymus lanceolatus		0.0	2	**2.6**		0.0	2	1.4
Hesperostipa comata		0.0		0.0	2	**7.1**	2	1.4
Linum perenne	1	**2.3**		0.0		0.0	1	0.7
Penstemon palmeri		0.0	15	**19.7**		0.0	15	10.2
Pleuraphis jamesii		0.0		0.0	9	**32.1**	9	6.1
Poa secunda		0.0	27	**35.5**	2	**7.1**	29	19.7
Sanguisorba minor	5	**11.6**		0.0		0.0	5	3.4
Sporobolus cryptandrus	6	**14.0**		0.0		0.0	6	4.1
N (species)	4		7		5		12	

Table 5-4. Incidence of twelve species seeded in four different vegetation types that burned in the Southern Nevada Complex fires of 2005. *Seeded* indicates plots where the species were intentionally seeded. The value 1 indicates the vegetation types where a species was present, with values in bold indicating the vegetation type where the species was intentionally seeded. A value of 0 indicates the species was seeded into that vegetation type but was not recorded in the BB plots. Vegetation types are: NRG = "natural regeneration" (creosote scrub and thermic blackbrush scrub, MBB = mesic blackbrush, NWPJ = non-wilderness pinyon-juniper, and WPJ = wilderness pinyon-juniper. NRG was not seeded.

Species	**Vegetation**			
	NRG	**MBB**	**NWPJ**	**WPJ**
Seeded				
Achnatherum hymenoides		**1**	**1**	**1**
Agropyron cristatum			**1**	
Agropyron fragile			**1**	
Elymus elymoides		0	**1**	**1**
Elymus lanceolatus			**1**	
Hesperostipa comata			0	**1**
Linum perenne		**1**		
Penstemon palmeri			**1**	0
Pleuraphis jamesii		0	0	1
Poa secunda		0	**1**	1
Sanguisorba minor		**1**		
Sporobolus cryptandrus		**1**	0	0
Unseeded				
Achnatherum hymenoides	1	1	1	1
Agropyron cristatum			1	
Agropyron fragile			1	
Elymus elymoides	1		1	1
Elymus lanceolatus				
Hesperostipa comata				1
Linum perenne				
Penstemon palmeri			1	
Pleuraphis jamesii	1			1
Poa secunda			1	1
Sanguisorba minor				
Sporobolus cryptandrus	1	1	1	1

Table 5-5. Mean stem density (stems per m^2 ± SE) of seeded species in seeded and unseeded treatment conditions in three vegetation types in the Southern Nevada Complex fires of 2005. Vegetation types are: NWPJ = non-wilderness pinyon-juniper, and WPJ = wilderness pinyon-juniper.

	Year			
	2006	**2007**	**2008**	**Mean**
Blackbrush				
Seeded	0.105 (0.010)	0.040 (0.033)	0.087 (0.033)	0.080 (0.028)
Unseeded	0.014 (0.001)	0.003 (0.003)	0.007 (0.007)	0.008 (0.004)
Mean	0.061 (0.004)	0.022 (0.017)	0.047 (0.018)	0.045 (0.015)
NWPJ				
Seeded	0.225 (0.078)	0.449 (0.121)	0.581 (0.167)	0.409 (0.071)
Unseeded	0.141 (0.057)	0.110 (0.032)	0.173 (0.036)	0.141 (0.025)
Mean	0.183 (0.048)	0.279 (0.064)	0.373 (0.086)	0.274 (0.038)
WPJ				
Seeded	0.356 (0.250)	0.458 (0.314)	0.254 (0.083)	0.356 (0.135)
Unseeded	0.054 (0.028)	0.070 (0.029)	0.338 (0.225)	0.155 (0.078)
Mean	0.208 (0.129)	0.268 (0.162)	0.296 (0.119)	0.257 (0.079)

Table 5-6. Mean stem density (stems per m^2 ± SE) of seeded herbaceous species in four areas that burned in the Southern Nevada Complex fires of 2005.

	Year			
	2006	**2007**	**2008**	**Mean**
Delamar				
Seeded	0.059 (0.030)	0.319 (0.111)	0.504 (0.205)	0.277 (0.074)
Unseeded	0.035 (0.013)	0.063 (0.021)	0.063 (0.020)	0.053 (0.010)
Mean	0.045 (0.015)	0.167 (0.049)	0.230 (0.082)	0.143 (0.031)
Duzak				
Seeded	0.174 (0.066)	0.203 (0.080)	0.176 (0.040)	0.184 (0.037)
Unseeded	0.089 (0.043)	0.063 (0.022)	0.099 (0.025)	0.084 (0.018)
Mean	0.129 (0.039)	0.126 (0.038)	0.133 (0.023)	0.129 (0.020)
Halfway				
Seeded	0.033 (0.033)	0.200 (0.100)	0.067 (0.067)	0.100 (0.044)
Unseeded	0.014 (0.014)	0.020 (0.014)	0.027 (0.018)	0.022 (0.010)
Mean	0.020 (0.013)	0.050 (0.025)	0.033 (0.018)	0.037 (0.012)
Meadow Valley				
Seeded	0.653 (0.336)	0.967 (0.472)	1.059 (0.465)	0.885 (0.242)
Unseeded	0.070 (0.046)	0.048 (0.032)	0.283 (0.181)	0.135 (0.064)
Mean	0.296 (0.137)	0.400 (0.190)	0.564 (0.208)	0.418 (0.104)

Although density of the seeded species was low, there was some evidence that aerial seeding increased establishment of the seeded species. The log-linear analysis indicated that the proportion of seeded species was greater in seeded than unseeded plots, and in pinyon-juniper than mesic blackbrush plots (Table 5-7 and Figure 5-2). However, seeding and year were minor effects, and although removal of these factors significantly reduced model fit, the reduction was relatively small (Table 5-7). Nonetheless, the best supported multi-level models for absolute density ($\log_e$ transformed) and relative density (after angular transformation) of the seeded species included seeding treatment (Appendix 5-1). Plots of absolute density of seeded species also indicated that they were higher in seeded than unseeded plots (Figure 5-3), although untransformed percent density data indicated little separation between seeded and unseeded conditions (Figure 5-4). Variation over time for the three response variables was minimal, as indicated by non-significant parameter estimates for year and year^2 in all of the models (Table 5-8), suggesting that at least during the first 3 years, there was no indication that the seeded and unseeded plots were diverging. Rather, variation for all four response variables was greatest among the vegetation types.

Table 5-7. Log-linear analysis of the frequency of seeded species from 2006 – 2008 in seeded and unseeded plots within three vegetation types in the Southern Nevada Complex fires of 2005. "Model without effect" refers to the fit of a simpler model (the effect and its interactions are removed but all other effects are included; a significant χ^2 value indicates that model fit is worse without the effect), while "Removal of effect" indicates if the reduction in fit is significant.

		Model without effect			Removal of effect		
Effect	$\log_e$ (MLE)	χ^2	*df*	*P*	χ^2	*df*	*P*
Vegetation	-183.83	193.54	12	0.000	191.10	8	0.000
Year	-46.51	18.89	12	0.091	16.45	8	0.036
Seeded	-45.09	16.06	9	0.066	13.63	5	0.018

In contrast to the effects of seeded vs. unseeded conditions, effects of vegetation type was by far the most influential in this study. Specifically, density (both absolute and relative) and diversity of the seeded species were significantly greater in the two pinyon-juniper types than mesic blackbrush (Table 5-8). These responses were very consistent, with < 7% of the variation being attributable to different patterns across the four fires (Table 5-8).

It is important to point out that great care must be made interpreting the analyses from the multi-level models. The residuals for both absolute and relative density were extremely non-normal and sometimes heteroscedastic, even after transformation. This may have resulted in bias in the parameter estimates, potentially making both the AIC and *P* values of these models suspect. Moreover, regardless of whether or not one considers the differences between seeded and unseeded plots to be statistically significant, the ultimate question relates to their ecological significance. Densities of seeded species across both seeded and unseeded plots were much lower than the threshold of desired densities specified in the Burned Area

Emergency Response plan for the 2005 Southern Nevada Complex (see limit lines in Figure 5-3). These issues at discussed at greater length in the following section.

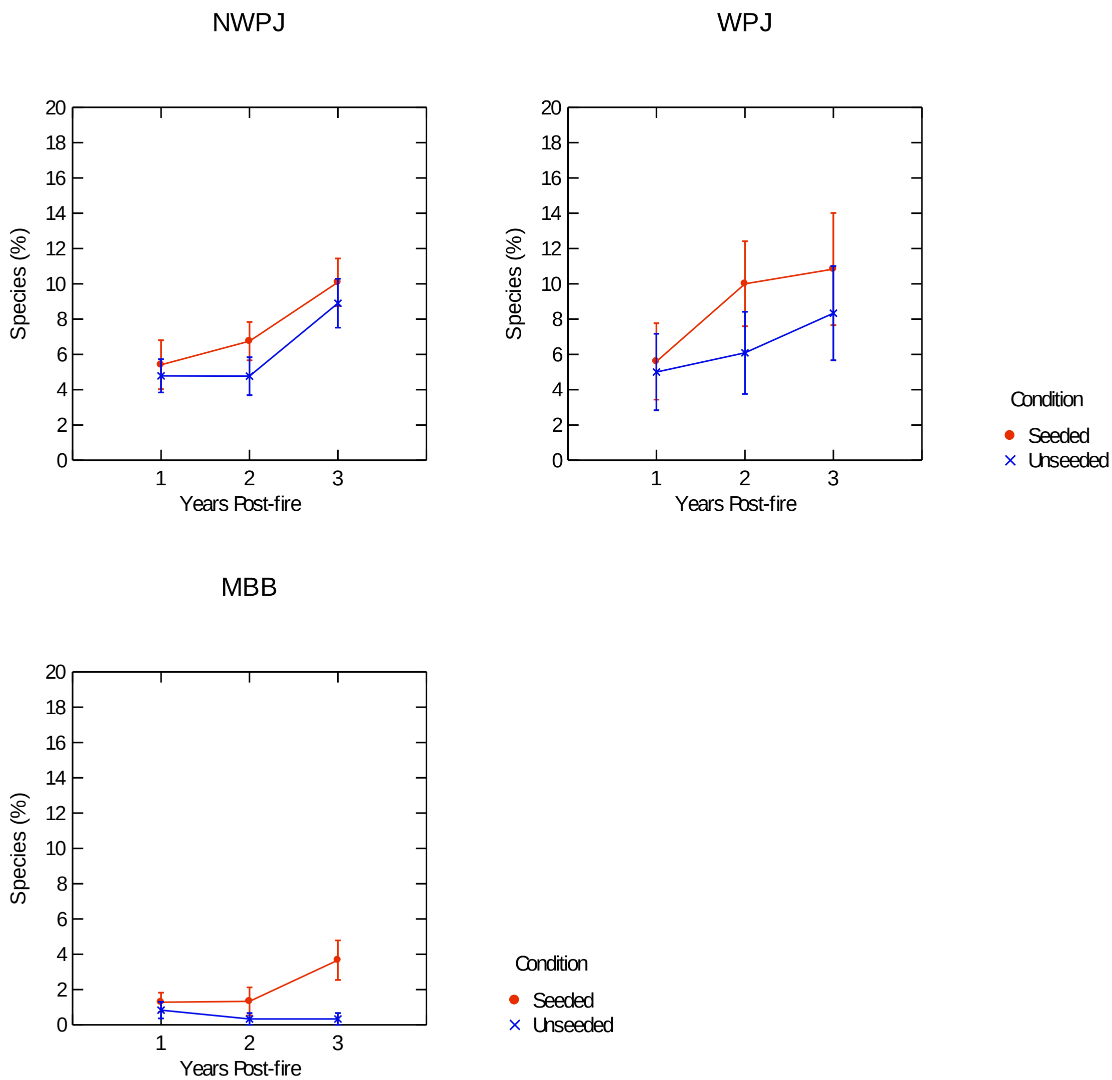

Figure 5-2. The mean percentage of seeded species in three vegetation types for seeded and unseeded conditions in the Southern Nevada Complex fires of 2005. Vegetation types were mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ).

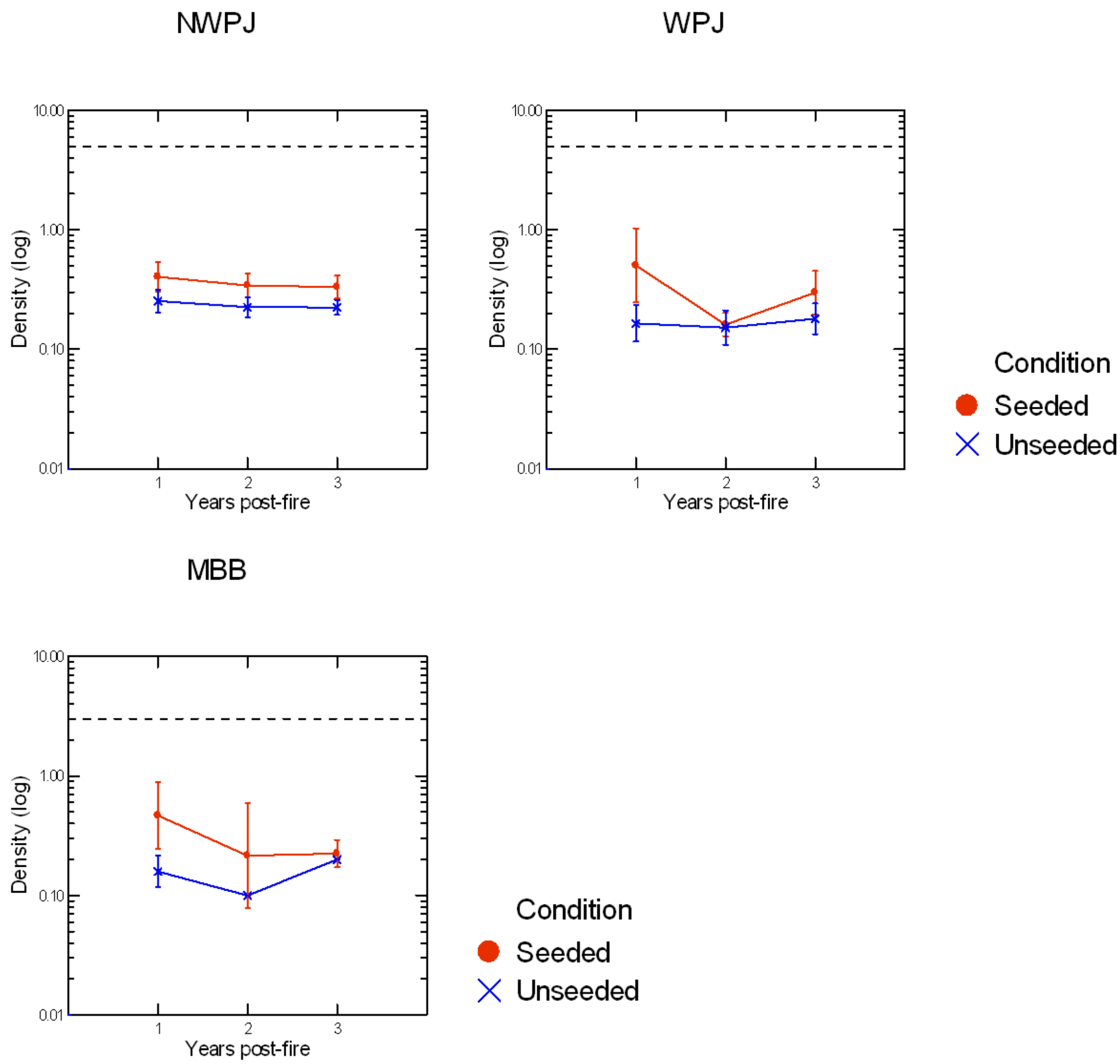

Figure 5-3. The mean density (individuals m^{-2}) of seeded species in three vegetation types for seeded and unseeded conditions in the Southern Nevada Complex fires of 2005. Vegetation types were mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The limit lines are threshold levels where aerial seeding is considered successful in the 2005 Southern Nevada Complex Burned Area Emergency Response plan (5 for non-wilderness and wilderness pinyon-juniper and 3 for mesic blackbrush).

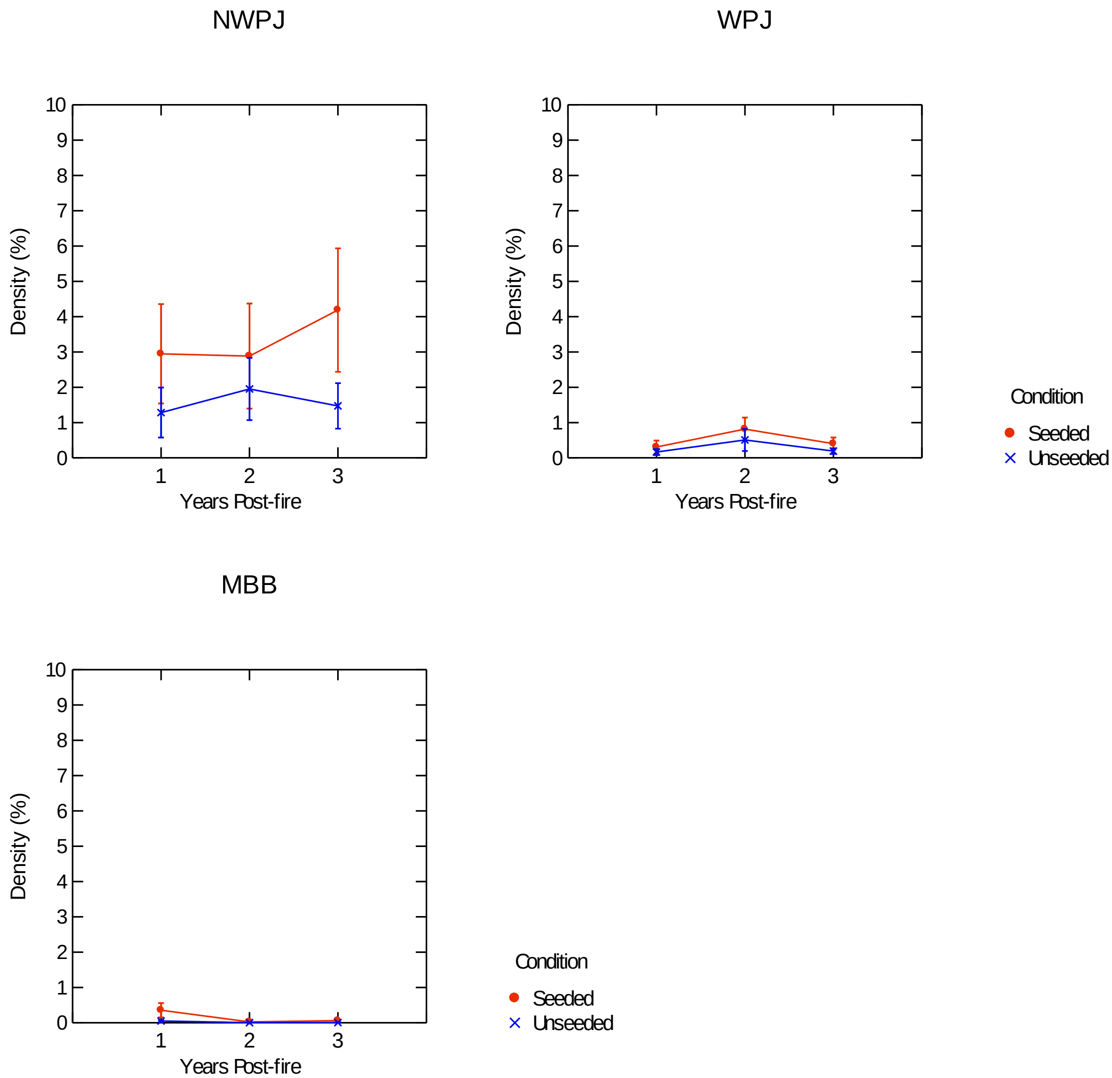

Figure 5-4. Relative density (percentage of individuals m^{-2}) of seeded species in three vegetation types for seeded and unseeded conditions in the Southern Nevada Complex fires of 2005. Vegetation types were mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ).

Table 5-8. Parameter estimates of three response variables in the best supported models of seeded species establishment in four fires (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Parameters for the non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) vegetation types are relative to that of mesic blackbrush. Fire, Plot, and Plot x years are random effects. The form of Simpson's index (*Simpson*) was $-\log_e$ (D), where D = Simpson's index of concentration.

Source of variation	Parameter	S.E.	Z	P	Variation (%)
Simpson					
Fixed factors					
Constant	-0.043	0.053	-0.811	0.791	
Year	0.016	0.059	0.271	0.393	
Year^2	0.003	0.015	0.200	0.421	
NWPJ	0.065	0.016	4.063	0.000	
WPJ	0.075	0.020	3.750	0.000	
Seeded	0.021	0.014	1.500	0.067	
Random factors					
Fire	0.000	0.001	0.000	1.0000	0.3
Error	0.034	0.002	17.000	0.0000	99.7
Seed Density (log)					
Fixed factors					
Constant	-0.056	0.087	-0.644	0.740	
Year	0.012	0.090	0.133	0.447	
Year^2	0.006	0.022	0.273	0.392	
NWPJ	0.136	0.029	4.690	0.000	
WPJ	0.082	0.033	2.485	0.006	
Seeded	0.095	0.021	4.524	0.000	
Random factors					
Fire	0.003	0.003	0.000	1.0000	3.7
Error	0.079	0.004	19.750	0.0000	96.3
Seed Density (%-arcsin)					
Fixed factors					
Constant	-0.020	0.039	-0.513	0.696	
Year	0.031	0.038	0.816	0.207	
Year^2	-0.005	0.009	-0.556	0.711	
NWPJ	0.055	0.012	4.583	0.000	
WPJ	0.016	0.014	1.143	0.126	
Seeded	0.024	0.009	2.667	0.004	
Random factors					
Fire	0.001	0.001	0.000	1.0000	6.7
Plots	0.014	0.001	14.000	0.0000	93.3

DISCUSSION

This study provides limited statistical evidence that seeded species occurred more frequently and at higher densities in seeded than unseeded plots. However, as discussed in the Results section of this report, many of the analyses were ambiguous and may be compromised by violations of the assumptions underlying the multilevel statistical models. Consequently, our confidence in the statistical validity of these analyses is tenuous. Support for relative density of the seeded species being greater in the seeded plots was even weaker, and there was no support for a consistent increase in density over time. Relative density is a particularly important variable because it reflects whether changes in absolute density were primarily by the seeded species or simply reflected broader, community-wide patterns. Frequency and density of seeded species was greatest in pinyon-juniper communities, especially non-wilderness pinyon-juniper, compared to mesic blackbrush. This was not restricted to seeded plots though; rather, it was an overall effect of vegetation type.

It is critical to note that a very clear distinction needs to be made between statistical and ecological significance. Even if the results of the analyses were less ambiguous and there was greater confidence that the results were robust to violations of statistical assumptions, the importance of any detectable statistical patterns in the seeding treatments is far less important than the detection of ecological patterns. Ecologically, the seeded species comprised a minor component of the post-fire vegetation communities over the three year duration of the study. Even if the number of species and their densities were considered to be higher in seeded than unseeded plots, their absolute values were very low. On average, densities of seeded species were 2-3 orders of magnitude lower than the desired levels of three seeded perennial species m^{-2} in mesic blackbrush and five seeded perennial species m^{-2} in pinyon-juniper that were listed in the Burned Area Emergency Response plan for the 2005 Southern Nevada Complex. Moreover, seeded species barely made up 1% of the relative abundance of all herbaceous species. Despite all of this, the patterns we report highlight useful questions and considerations that could help improve future management of post-fire communities in the Mojave bioregion.

A basic question has been raised regarding the need for aerial seeding following fires. There is ongoing research in Mediterranean ecosystems of North America on the success of aerial seeding for stabilizing soil relative to causing other effects, such as increased propagule pressure of non-native species (Beyers 2004, Keeley et al. 2006). These questions and research programs are equally applicable to arid ecosystems such as the Mojave, where systematic evaluations of the success of aerial seeding treatments are surprisingly limited. The results of our analyses do not necessarily indicate that all aerial seeding treatments in the blackbrush and pinyon-juniper zones of the Mojave Desert will have low establishment rates, and in fact BLM Ely District files indicate that seeding objectives have been achieved for

some postfire seeding projects in the past. However, they do indicate that systematic evaluations of the treatments should be conducted more frequently and should include empirical comparisons with untreated control areas.

A particularly important goal of future seeding treatments should be a more systematic approach to identify modifications that would increase their likelihood of success. Post-fire aerial seeding in the Mojave bioregion is currently being conducted with substantial variability in timing, seed mixes, application rates, and the conditions where seeding is done (e.g. vegetation types). This variability, as well as the limited data, makes it difficult to place the outcomes of the seeding treatments in the Southern Nevada Complex fires in the broader context of other fires occurring in this region during other years.

A straightforward adjustment that could be made to the way future Mojave Desert seeding treatments are done is to place them in a framework that is more aligned with adaptive management philosophies. Rather than evaluating aerial seeding as successful or unsuccessful based on predetermined desired densities of established plants, a more systematic approach that identifies and evaluates the conditions under which post-fire aerial seeding would have the greatest likelihood of success would be very beneficial to managers. This approach could include the integration of monitoring, such as that conducted for this project, with field experiments focused on determining what time of year, rates, and species (including the number of species in a mix, their identity, and what proportion multiple species occur in) would be most likely to achieve management goals. A particularly important part of this approach would be determining if there are some conditions where seeding has a very low likelihood of success. For example, if a rapid post-fire assessment of species composition in the seed bank indicates that post-fire succession patterns are going to be dominated by non-native species, then it may be best to not allocate resources to aerial seeding treatments unless efforts to directly reduce dominance of non-native annuals (e.g. herbicides) are implemented first. Results presented in Chapter 6 of this report plus other published studies (e.g. Brooks 2000, DeFalco et al. 2006, Eiswerth et al. 2009) suggest that establishment of seeded plants may be exceedingly low in areas dominated by non-native annuals. An alternative to extensive postfire assessments might be predictive models that can be used to estimate where non-native annual plants are most likely to dominate, especially immediately following fire, and avoid seeding those areas. Bioclimatic models for *Bromus* have been developed in the Great Basin (Bradley 2009), and models relating burn severity with postfire dominance by *Bromus* have been developed in the Colorado Plateau (Ethan Aumack personal communication). Similar models would be useful if developed for the non-native annuals *Bromus rubens*, *Bromus tectorum*, *Schismus* spp. and *Erodium cicutarium* in the Mojave Desert.

The identification of the most appropriate species to use in seed mixes is an initial step that may improve seeding success as well as the justification for doing seeding treatments. Sixteen species were used in the seed mixes in this project, all of which had limited distributions and low abundance. However, several species appeared to have a much higher chance of establishment in at least one of the vegetation types evaluated in this study. These species included *Achnatherum hymenoides*, *Elymus elymoides*, *Pleuraphis jamesii*, and *Poa secunda* all of which are native species, and *Agropyron cristatum* and *Sanguisorba minor* which are non-native. The four native species are clearly candidates for systematic evaluations of their likelihood of establishment. However, there should be a careful evaluation of the value for continuing use of non-native species in seed mixes. One of the major criticisms of post-fire seeding is that the use of non-native species increases the likelihood of them dominating post-fire communities, competing with native species, and potentially causing unexpected ecosystem-level impacts (Keeley 2006, Keeley et al. 2006, Klinger et al. 2006, 2008). That said, one of the primary reasons these two non-native species are included in seed mixes is because they are believed to compete with non-native annual grasses and subsequently reduce the frequency of fire. There is some evidence that this is true for *Agropyron cristatum* competing with *Bromus tectorum* in the Great Basin (e.g. Eiswerth et al. 2009), but these relationships have not been substantiated in the Mojave Desert. Abundances of *Sanguisorba minor* and *Bromus* spp. have been observed to be somewhat mutually exclusive in the northwestern Mojave Desert (K. Prentice personal observation), and the inference is that there may be competition occurring between the two species. This thinking has been one of the reasons that *Sanguisorba minor* has been included in post-fire seed mixes in the northeastern Mojave Desert, although there are no studies to support this justification. Any time a non-native species is included in a seed mix, its potential ecological benefits must be weighed against the potential detriments. This evaluation process must also consider the reliability of available information needed to make these decisions. Thus, we believe specific studies should be conducted to evaluate the ultimate effects of seeding *Agropyron cristatum* and *Sanguisorba minor*, or any other non-native species, in the Mojave Desert.

Serious consideration should also be given to seeding not just with perennial herbaceous species, but also with a wider variety of woody species. Some vegetation communities in the Mojave are intolerant of fire, so seeding these communities with woody species could increase rates of woody cover recovery and, potentially, over time reduce abundance of non-native annual grasses and forbs. In addition, application of annual grass-specific herbicides prior to seedings might help to reduce competition of the seeded species with established *Bromus* spp. and *Schismus* spp. plants, at least during the first growing season.

A further modification to seeding treatments worth exploring is the pattern and extent of seeding. Currently, many treatments are done over relatively large, contiguous areas, and following the 2005 Southern Nevada Complex seeding polygons ranged from approximately 100 to 5,500 acres. The likelihood of success may potentially be enhanced if multiple seeding efforts in many smaller areas were used to establish "nascent foci" (*sensu* Moody and Mack 1988; also see Leger 2008) that, over time, could serve as colonizing sources for propagules of seeded species into the surrounding landscape. This would not be appropriate in treatments focused on soil stabilization, but rather for improving establishment of native perennials. Specific guidelines such as ideal size, shape, and distance between seeded patches are not currently available and should be the focus of further research efforts.

We do not believe that the results presented in this chapter were artifacts of poor study design, biased sampling, or that additional sampling would lead to a different interpretation of the outcomes of the seeding. We obtained GPS flight lines from the aerial seeding applications to ensure that none of them overlapped the unseeded control plots. Although the recorded flight lines indicated that the aircraft did not achieve 100% accuracy in their seeding applications, and in a few cases flight lines passed over designated control plots, the vast majority of the flight lines did not pass over areas means to be unseeded controls. In addition, sample sizes were quite large (Chapter 4), and the monitoring design and sampling protocols were appropriate for the questions being asked. Pre-treatment data is always desirable in intervention projects because they serve as a baseline to evaluate changes relative to background levels, but being able to compare seeded with unseeded plots helped to compensate for the lack of pre-seeding data. The three-year duration of this project was probably not long enough to detect the establishment of adult plants from seedings, and it is possible that continued monitoring may detect establishment in the long-run. However, the vegetation throughout most of the burned areas is dominated by non-native annual grasses and forbs (see Chapter 6), and these species seem to be exerting such a strong competitive effect on other plants that it is difficult to imagine that germination, establishment, and growth of the remaining seeds that may still reside in the seedbank over the next 3-5 years or longer would be enough to change this situation. In addition, initial seeding rates were relatively low, and after 3 years there may be few seeds left ungerminated in the soil seedbank. However, there may also be detectable changes in cover over time as some of the perennial grasses that were seeded and established grow to maturity, assuming they are not outcompeted by non-native grasses during their early life stages. Accordingly, additional monitoring of these plots to evaluate trends in cover would be of value.

There are also many situations where data collected over periods of time longer than 3 years would be extremely effective at detecting temporal trends in establishment of seeded species. This would be especially so for vegetation communities where non-native herbaceous species do not dominate succession patterns, and if woody species are used in

seed mixes. Serious consideration should be given by BLM to adopt a design similar to that used by the National Park Service, where monitoring data following fuels treatments (including wildland fire use fires) is collected at intervals of 1, 2, 3, 5, 7, 10, and 15 years (USDI National Park Service 2001).

Postfire aerial seeding is typically done to suppress dominance by undesirable non-native annual grass species (e.g. cheatgrass *Bromus tectorum*, red brome *Bromus rubens*, Mediterranean split grass *Schismus* spp.) and/or stabilize soils. The results presented in this chapter and Chapter 6 of this report indicate that aerial seeding in the 2005 Southern Nevada Complex fires most likely had no effect on suppression of undesirable non-native species in the burned areas within the mesic blackbrush and pinyon-juniper zones. In many places seeds of *Bromus* spp. were observed to cover the ground following the fires and appeared to have survived unburned (M. Brooks personal observation). These surviving seeds likely contributed to the masses of non-native seedlings that appeared by the time aerial seedings were applied the winter following the fires. In some places within the mesic blackbrush and upper creosotebush ecotones these non-native plants were already present in biomass levels approaching 500 lbs/acre (560 kg/ha) during February 2006 (M. Brooks unpublished data). Counter to management goals, it is likely that competition from these previously established non-native annuals suppressed germination, growth, and establishment of the species that were seeded over them, and not the other way around. In addition, while not desirable from a conservation perspective, the density of non-native annual species was several orders of magnitude greater than that of the seeded species (see Chapter 6), which likely negated any immediate need for aerial seeding to achieve soil stabilization objectives (see Chapter 8).

MANAGEMENT IMPLICATIONS

This chapter reports marginally significant establishment rates of seeded species during the first 3 postfire years following the 2005 Southern Nevada Fire Complex. However, their absolute densities were far below the objectives stated in the seeding plans, and at levels that may be ecologically insignificant. This calls into question the value of the aerial seeding treatments that were applied. Although the long-term effects of the seedings may not yet be realized, the prevalence of non-native annuals across the burned areas and their negative correlations with establishment rates of perennial plant seedling (Chapter 6) suggests that additional recruitment of seeded species past the initial sampling period of 3 years may be negligible. Periodic monitoring of the sampling plots during subsequent years should be done to test this hypothesis, perhaps during postfire years 5, 7, 10, and 15.

The results reported in this chapter relate to a specific set of postfire seeding treatments applied during a single year. Variable characteristics among seeding treatments and the vagaries of climatic conditions make it difficult to generalize the results of these

seedings to other years. More information is clearly needed on how establishment rates vary with the seasonal timing of seeding, species composition of seed mixes, and application rates, and among vegetation types and years of contrasting climatic conditions. This can be accomplished by monitoring the establishment rates of future postfire seeding treatments, but only if replication of seeding treatments and unseeded controls are included in the monitoring plans. However, this haphazard approach would take much longer to produce comprehensive results than a series of well-designed experiments that manipulate the various variables so that a predictive model can be developed.

More information is also needed beyond just understanding how to get seeds to establish and grow into mature plants. Specifically, information is needed on the potential for seeding treatments to achieve their ultimate objectives. If the objectives are to reduce soil erosion, then there may be some cases where seedings are warranted, specifically of perennial species at higher elevations in pinyon-juniper vegetation (see Chapter 8). In contrast, if the seeding objectives are to reduce dominance of non-native annual plants, then the monitoring results for the 2005 Southern Nevada Complex are much less supportive (see Chapter 6 and the current chapter). Specifically, establishment of perennial species seedlings only occurred where rainfall levels were high and non-native annuals densities were low (see Chapter 6), which suggests that seedings only be done where non-native abundances are low. In addition, densities of native and non-native annuals were actually positively correlated, providing no evidence for competition between the two, thus providing no justification for seeding annual forbs to compete with non-native annual grasses in future seeding projects. There may of course be other reasons for seeding post-fire landscapes, such as to provide forage and cover vegetation for specific species such as the desert tortoise (*Gopherus agassizii*). In these cases annual forage species may be able to establish (see desert tortoise seeding results in Chapter 3 of this report), but the establishment of perennial cover species may only be expected under the limited window of opportunity of high rainfall and low non-native annual density conditions discussed above.

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Appendix 5-1. Model selection statistics for three response variables of seeded species establishment in four areas (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to mesic blackbrush. AICc is the bias-corrected Akaike Information Criterion, ΔAICc is the difference in AICc between a given model and the best supported model, exp(ΔAICc) is the absolute support for a given model, and wAICc is the support relative to the other models.

Model	**Variables**	**AICc**	**ΔAICc**	**exp(ΔAICc)**	**w AICc**
Simpson					
6	Model 5 + seeded	-375.16	0.00	1.0000	0.4412
5	Model 4 + vegetation type	-374.84	0.31	0.8544	0.3770
7	Model 6 + year*vegetation type	-372.67	2.49	0.2884	0.1272
8	Model 7 + year*seeded	-370.65	4.51	0.1048	0.0463
9	Model 8 + vegetation type*seeded	-366.60	8.55	0.0139	0.0061
10	Model 9 + $year^2$*vegetation type	-362.55	12.61	0.0018	0.0008
3	Model 2 + year	-362.34	12.82	0.0016	0.0007
11	Model 10 + $year^2$*seeded	-360.46	14.70	0.0006	0.0003
4	Model 3 + $year^2$	-360.33	14.83	0.0006	0.0003
12	Model 11 + $year^2$*NWPJ*seeded	-357.59	17.57	0.0002	0.0001
1	Null - fixed intercept	-355.56	19.60	0.0001	0.0000
2	Random intercept - fire	-353.95	21.21	0.0000	0.0000
13	Model 12 + $year^2$*WPJ*seeded	-353.42	21.73	0.0000	0.0000
Seed Density (log)					
6	Model 5 + seeded	235.35	0.00	1.0000	0.5045
7	Model 6 + year*vegetation type	236.58	1.23	0.5398	0.2723
8	Model 7 + year*seeded	237.71	2.36	0.3067	0.1547
9	Model 8 + vegetation type*seeded	239.82	4.47	0.1072	0.0541
10	Model 9 + $year^2$*vegetation type	243.75	8.40	0.0150	0.0076
11	Model 10 + $year^2$*seeded	244.93	9.58	0.0083	0.0042
12	Model 11 + $year^2$*NWPJ*seeded	246.23	10.87	0.0044	0.0022
13	Model 12 + $year^2$*WPJ*seeded	249.42	14.07	0.0009	0.0004
5	Model 4 + vegetation type	253.65	18.30	0.0001	0.0001
3	Model 2 + year	268.62	33.27	0.0000	0.0000
4	Model 3 + $year^2$	270.63	35.28	0.0000	0.0000
2	Random intercept - fire	273.40	38.05	0.0000	0.0000
1	Null - fixed intercept	285.18	49.83	0.0000	0.0000

Appendix 5-1 continued.

Model	Variables	AICc	ΔAICc	exp(ΔAICc)	w AICc
Seed Density (%-arcsin)					
6	Model 5 + seeded	-997.59	0.00	1.0000	0.5179
7	Model 6 + year*vegetation type	-996.29	1.30	0.5223	0.2705
8	Model 7 + year*seeded	-994.74	2.85	0.2405	0.1246
5	Model 4 + vegetation type	-992.23	5.36	0.0686	0.0355
9	Model 8 + vegetation type*seeded	-992.05	5.54	0.0627	0.0325
10	Model 9 + $year^2$*vegetation type	-990.24	7.35	0.0254	0.0131
11	Model 10 + $year^2$*seeded	-988.27	9.32	0.0095	0.0049
12	Model 11 + $year^2$*NWPJ*seeded	-984.62	12.97	0.0015	0.0008
13	Model 12 + $year^2$*WPJ*seeded	-982.05	15.54	0.0004	0.0002
3	Model 2 + year	-976.67	20.92	0.0000	0.0000
4	Model 3 + $year^2$	-975.01	22.58	0.0000	0.0000
2	Random intercept - fire	-974.31	23.28	0.0000	0.0000
1	Null - fixed intercept	-937.56	60.03	0.0000	0.0000

Chapter 6: Vegetation Trends Following the 2005 Southern Nevada Fire Complex

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INTRODUCTION

Fire was long assumed to be a relatively uncommon event in the Mojave Desert (Humphrey 1974). This perspective was due in part to the general observation that arid, low productivity ecosystems have fewer fire events than more productive ecosystems. It is also influenced by the fact that much of the Mojave Desert is dominated by sparse low elevation shrublands of discontinuous fuels which historically resulted in relatively infrequent fires and a low proportion of area burned (Brooks and Matchett 2006, Brooks and Minnich 2006). The occurrence of fire was greater in mid and upper elevation zones within the bioregion, where fuels were denser and more continuous. In addition, over the last thirty years there has been a major increase in cover of non-native annual herbaceous species in virtually all elevation zones in the Mojave Desert (Brooks 1999, Brooks and Berry 2006), leading to increased fire sizes, especially within the lower and middle elevation zones (Brooks and Matchett 2006). This has been especially so during wetter years, when higher productivity and biomass of non-native annual species increases the probability of ignition and fire size, as well as facilitating fire spread (Brooks and Matchett 2003, 2006).

Although it is now recognized that fire regimes are shifting in many areas of the Mojave Desert (Brooks and Esque 2002, Brooks and Matchett 2006, Brooks and Minnich 2006), variations in post-fire succession patterns across the landscape and over time are still not well understood. Relatively few woody species in the Mojave Desert are fire-tolerant, so initial regeneration rates from resprouts and seedlings are often low (Brooks and Matchett 2003, Brown and Minnich 1986, West and Young 2000). However, some species can be very resilient to fire, possibly due to evolutionary histories that may have included physical disturbances such as flooding or even historical fire at higher elevations. In addition, there is widespread concern that a grass-fire cycle (D'Antonio and Vitousek 1992) has become established in the Mojave (Esque and Schwalbe 2002), but there is also evidence that it is not occurring everywhere (Brooks and Esque 2002, Brooks and Matchett 2006). The grass-fire cycle is a feedback system where an initial disturbance or series of disturbances (not necessarily fire) facilitates increased abundance of non-native annual grasses, which then facilitate recurrent fire. This results in a greater likelihood of large fires and short fire return intervals, which in turn leads to high mortality of seedling and sapling shrubs. Non-native annual grasses are able to persist in systems with short fire return intervals, thereby perpetuating the cycle. Over time, the grass-fire cycle can lead to the complete transformation of shrub communities to communities dominated by non-native annual grasses.

In this chapter we analyze the spatial and temporal succession patterns in the Delamar, Duzak, Halfway, and Meadow Valley fires of 2005, especially in relation to dominance by non-native grasses and forbs. We focus particularly on distribution and

abundance patterns of cheatgrass (*Bromus tectorum*), red brome (*Bromus rubens*), and red-stemmed filaree (*Erodium cicutarium*). It has been recognized for many decades that cheatgrass has played a key role in the transformation of a range of ecological characteristics in the Great Basin bioregion (Stewart and Hull 1949). This includes alterations to the structure of vegetation communities (e.g. a much higher proportion of annual cover, especially in spaces between shrubs), species composition (reduced proportion of native cover), and ecosystem processes (e.g. water availability, fire regimes) (Young and Clements 2009). Cheatgrass increases the continuity of fine fuels, which in turn increases the probability of ignition and hence fire frequency (Whisenant 1990). It is estimated that, over the last century, fire frequency in the Great Basin has increased from approximately once every 30 to 110 years to once a decade or less (Whisenant 1990, Wright and Bailey 1982). By the 1990's it was observed that the grass-fire cycle was becoming more prevalent in the Mojave Desert as well (Brooks and Pyke 2001). However, in the Mojave it has been attributed to increased abundance of red brome and two species of Mediterranean split grass (*Schismus arabicus* and *S. barbatus*). Observations indicate that cheatgrass is more abundant in mid and higher elevation communities of the Mojave (blackbrush, interior chaparral, and pinyon-juniper woodland), red brome in lower and mid-elevation communities (creosote scrub and blackbrush), and the two *Schismus* species in low elevation communities (creosote scrub) (Brooks and Berry 2006). In addition to the non-native annual grasses, *Erodium cicutarium* (*Erodium* from hereon) is frequently a dominant component of the ground cover in many mid to low elevation communities in the Mojave (Brooks and Berry 2006, especially in post-fire communities (Brooks and Matchett 2003). Although *Erodium* does not promote fire in the way that non-native annual grasses do (Brooks 1999), there is evidence that it can alter vegetation structure and, potentially, post-fire succession patterns (Brooks and Matchett 2003).

Our specific objectives in this chapter were to analyze: (1) the relationship of vegetation structure and species composition with topographic, disturbance, and rainfall variables, (2) the spatial and temporal patterns of abundance of cheatgrass, red brome, and *Erodium* in different vegetation communities, as well as their relationships to gradients in elevation, precipitation, and disturbance, and, (3) the correlation between patterns of native herbaceous and woody species with abundance of cheatgrass, red brome, and *Erodium*, and discuss how this is likely determining succession trajectories in these post-fire communities.

METHODS

Site Description and Study Design

In this chapter we describe postfire successional patterns in the Delamar, Duzak, Halfway, and Meadow Valley fires of the Southern Nevada Fire Complex of 2005. We focused our analyses on four vegetation types that are arranged along an elevation gradient in the Mojave

Desert. Starting at the highest elevations are the wilderness and non-wilderness pinyon-juniper vegetation categories, then at middle elevations is the mesic blackbrush vegetation type, and at the lower elevations is the “natural regeneration” vegetation type. The latter vegetation type includes both thermic blackbrush and upper elevation creosotebush scrub, and is referred to as natural regeneration in this chapter because that is how it is referenced in the BAER plan and in other documents associated with the postfire management of the Southern Nevada Fire Complex of 2005. Those thermic blackbrush and upper elevation creosotebush areas were assigned the “natural regeneration” treatment, which meant that the treatment prescription was not to apply seeding or other direct management treatments, but rather to let them recover on their own. Chapters 1 and 4 of this report contains more details about the 2005 Southern Nevada Complex fires, the characteristics of the areas that were monitored, the vegetation sampling design, and the sampling schedule. In the current chapter we only use data from the BB plots. As described in Chapter 5 of this report, these were the most ecologically meaningful and most consistently collected data.

Our general approach was to first assess succession patterns in diversity and species composition of herbaceous and woody species (sub-shrubs, shrubs, and trees), then focus on patterns for six guilds of plants: native annual and perennial forbs, native annual and perennial grasses, and non-native grasses and forbs. Non-native annuals and perennials were pooled because annuals dominated the species composition and abundance, which means that our analyses are essentially evaluating patterns of annual species (see Results below). In addition to the guilds, we also analyzed the response of cheatgrass, red brome, and *Erodium*. Because seeding effects were negligible (see Chapter 5), seeding was not included as a factor in the analyses in the current chapter.

Data Analysis

Community Composition. We used Canonical Correspondence Analysis (CCA; ter Braak 1995) to evaluate spatial and temporal changes in species composition. Separate analyses were done for herbaceous and woody species using log + 1 transformed data. The environmental variables included three topographic variables (elevation, slope, and aspect), an index of disturbance amount (fire severity as measured by RdNBR, the relative differenced normalized burn ratio; Miller and Thode 2007), and five precipitation variables (centimeters of rain in five seasons: monsoon, late dry, early wet, late wet, and early dry). Aspect was arcsine transformed. Monsoon included August and September in the year prior to sampling, the late dry October and November, early wet December through February, late wet March and April, and the early dry May through July. RdNBR is roughly equivalent to the proportional amount of vegetation consumed by fire and is explained further in Chapter 9 of this report. Precipitation data were downloaded from the National Oceanic and Atmospheric (NOAA) National Climate Data Center and then extracted for each plot in each year with ArcGIS. The CCA’s were based on the linear combination of the environmental

variables and the species cover values (Palmer 1993). Forward stepping multiple regression was used to select the variables that contributed significantly to the ordination, and permutation tests were used to test the significance of the first ordination axis and the overall ordination (ter Braak 1995). A total of 199 permutations were used to calculate significance levels, with Fire included as a covariable (i.e. all permutations were partitioned within the four fires).

Community Structure. We derived four diversity indices for both herbaceous and woody species. Three of these indices comprised Hill's series (Hill 1973, Magurran 2004): N_0, the overall species richness in a sample; N_1, which equals $\exp^{H'}$, where H' is Shannon's index of diversity; and N_2, the reciprocal of Simpson's index (see Chapter 5). Hill's series is considered one of the most useful measures of diversity because the units are species numbers (more specifically, the "effective" number of species in a sample) (Routledge 1979, Tothmeresz 1995, Legendre and Legendre 1998, Magurran 2004). The fourth index we derived was Simpson's index of evenness:

$$E_{1/d} = (D^{-1}/S)$$

where D = Simpson's index of concentration (see Chapter 5) and S the total number of species in a sample. $E_{1/d}$ has what are generally considered to be the most desirable properties among evenness indices, especially because it is not sensitive to differences in species richness among samples (Magurran 2004, Smith and Wilson 1996). It is particularly useful when a community is dominated by a few species (see Results; Smith and Wilson 1996). The diversity indices for woody species were derived from density of resprouts and mature individuals.

Multi-level models were used to analyze general patterns of vegetation structure (density of herbaceous and woody species) and diversity (herbaceous and woody species), as well as abundance and species richness (absolute and relative) within each of the six herbaceous guilds. Multi-level models are a very flexible and robust set of methods based on maximum-likelihood estimates of parameters, rather than ordinary least square estimates such as used in rmANOVA (Gelman and Hill 2007). Because they are based on maximum likelihood estimates, multi-level models can handle situations where not all plots are sampled each year, which was the case in this project (conventional repeated-measures analysis of variance requires that all sample units in the analysis are sampled at each time occasion). Multi-level models also partition variance based on random and fixed factors. Predictor variables in the models included fixed and random effects, the linear and quadratic effects of time (year and year^2), vegetation type (natural regeneration, mesic blackbrush, non-wilderness pinyon-juniper, and wilderness pinyon-juniper), and the interactions between the time variables and vegetation type. Fire (Delamar, Duzak, Halfway, and Meadow Valley)

was the random factor in all of the models except the one for fixed effects (constants only model), which assumes variation is constant across fires. Akaike's Information Criteria (AIC) was used to determine the model with the greatest support from the pool of 7 possible models. Significance of parameters in the best model was tested with a Z-statistic (calculated by dividing the parameter estimate by its model-based standard error).

Red Brome, Cheatgrass, and Erodium Patterns. We used generalized linear models (GLM) to develop species response curves for cheatgrass, red brome, and *Erodium* along the gradient in elevation. The response curves were derived from counts of stems, so a logarithmic link function was used so that estimated values were constrained to an ecologically realistic range of zero or greater (equivalent to Poisson regression; McCullagh and Nelder 1989). We compared the relative fit of linear and unimodal response models with the AIC statistic.

We used GLM's with a Gaussian error structure and identity link (i.e. simple least square regression) to analyze the relationships between precipitation, RdNBR, and elevation. We then used GLM's with a Poisson error structure and log link to analyze the relationship of cheatgrass, red brome, and *Erodium* density with residuals for precipitation and RdNBR.

Multilevel models were used to analyze the relationship of the diversity indices, as well as density and species richness of the native herbaceous guilds and the density of woody stems and seedlings, with RdNBR (residuals), precipitation (residuals), and the density (log + 1 transformed) of cheatgrass, red brome, and *Erodium* (pooled).

Unburned Plots. We derived summary statistics on the absolute and relative species richness, density, and cover for six guilds of herbaceous plants in the 85 unburned plots. The guilds included native annual and perennial forbs, native annual and perennial grasses, and non-native annual grasses and forbs. Because these plots were sampled only in 2006 we did not include them in the analyses described above, but used them primarily as a reference to pre-burn conditions in the Delamar, Duzak, Halfway, and Meadow Valley fires.

Cover. Analysis of patterns based on cover data were complicated by the change in 2007 from point-intercept sampling to ocular estimates (see Chapter 4). Although both point-intercept and ocular estimate sampling was done at a subset of the plots in 2007 (N = 162), all but eight of these were in pinyon-juniper vegetation. Moreover, the accuracy of ocular estimates made at a scale of 150 m^2 are suspect (see Chapter 4). Therefore, we derived estimates of cover by relating cover values made with point-intercept sampling to stem density values. Cover and biomass would be expected to have a generally increasing relationship up to a threshold point, at which point the relationship would become asymptotic.

The analysis was conducted for all plots in 2006 and 2007 where cover estimates from point-intercept sampling and density estimates were available (N = 466). We used GLM's with a binomial error structure and logit link to analyze the relationship between percent cover and density ($\log_e + 1$ transformed) for five vegetation guilds: non-native annual grasses and forbs, native forbs (annuals and perennials combined), native grasses (annuals and perennials combined), and woody species (shrubs and trees). Native forbs and grasses were pooled into one guild because of the low cover values for annual and perennial species. Angular transformation was used for cover of native forbs, native grasses, and woody species. Predictor variables in the models included density, density^2, vegetation type, and their interaction. AIC was used to compare the models. The parameter estimates based on the 2006 and 2007 data were then used to derive cover estimates for each guild in those plots where point-intercept sampling was not conducted in 2007 and all plots sampled in 2008.

Differences in absolute cover across years and vegetation types were analyzed with multi-level models. Separate analyses were conducted for each guild. Predictor variables in the models included the linear and quadratic effects of time (year and year^2), vegetation type (natural regeneration, mesic blackbrush, non-wilderness pinyon-juniper, and wilderness pinyon-juniper), and the interactions between the time variables and vegetation type. Fire (Delamar, Duzak, Halfway, and Meadow Valley) was the random factor in all of the models except the one for fixed effects (constants only model), which assumes variation is constant across fires. AIC was used to compare the models.

RESULTS

Community Composition

All of the environmental variables provided significant explanatory value in the ordination of herbaceous species composition ($P \leq 0.014$) (Figure 6-1). These variables included topographic elevation, slope, and aspect, fire severity (RdNBR), and precipitation during the monsoon, late dry, early wet, late wet, and early dry seasons. Precipitation during the monsoon season was highly correlated with precipitation in the late wet season ($r = 0.759$), however the variance inflation factors (VIF) for monsoon and late wet season did not indicate multicollinearity (VIF = 5.33 for monsoon, 5.40 for late wet). Regardless, inclusion of both variables in the analysis did not provide unique information beyond that provided by just one, so monsoon was dropped from the analysis, which resulted in the VIF for late wet season dropping too 2.99.

Differences in herbaceous species composition for the first axis of the CCA and the overall ordination were both significant ($\lambda = 0.272$, $F = 58.27$, $P \leq 0.002$ for the first axis, $\lambda = 0.476$, $F = 13.41$, $P \leq 0.002$ for the overall ordination). The first axis was a primarily a

topographic gradient most influenced by the effects of elevation, slope, and RdNBR (Table 6-1). The second and third axes were gradients in the timing and amount of rainfall during the early versus late wet season (axis 2; 17.4% variation explained) and early versus late dry season (axis 3; 8.0% variation explained). Species composition of herbaceous species varied distinctly across the topographic gradient of the first axis from natural regeneration plots to mesic blackbrush plots to wilderness and non-wilderness pinyon-juniper plots as the plots progressed towards higher elevations, slopes, and burn severities (Figure 6-1A). Composition also clearly varied among years depending on the relative prevalence of early wet season precipitation versus late wet precipitation (Figure 6-1B). Individual herbaceous species and guilds of species also varied significantly along these same two CCA axes, with temporal shifts in spatial patterns occurring within each community in response to the timing and amount of rainfall (Figure 6-2).

Table 6-1. Correlation matrix and ordination statistics from a Canonical Correspondence Analysis of herbaceous species composition for four vegetation types burned in the southern Nevada Complex fires of 2005. λ is the eigenvalue (variance) for each axis, and RdNBR is an index of fire severity.

Variable	**Axis 1**	**Axis 2**	**Axis 3**
Elevation	0.941	0.153	0.052
Slope	0.596	0.061	-0.313
Aspect	0.274	0.084	-0.251
RdNBR	0.770	0.061	-0.059
Late dry season precipitation	0.208	0.126	-0.603
Early wet season precipitation	-0.045	0.817	-0.406
Late wet season precipitation	0.337	-0.877	-0.156
Early dry season precipitation	0.112	0.252	-0.582
	Axis 1	**Axis 2**	**Axis 3**
λ :	0.335	0.095	0.043
Species-environment correlations	0.841	0.642	0.527
Cumulative percentage variance			
species data	7.7	9.9	10.8
species-environment relation	61.7	79.1	87.1

All of the environmental variables also provided significant explanatory value in the ordination of woody species composition ($P \leq 0.018$) (Figure 6-3), but as with the herbaceous species ordination, precipitation during the monsoon season was highly correlated with precipitation in the late wet season (r = 0.811; VIF = 7.04). After monsoon was dropped from the analysis the VIF for late wet season dropped from 5.66 to 2.99. All other variables were retained in the analysis.

Differences in woody species composition for the first axis of the CCA and the overall ordination were both significant (λ = 0.464, F = 29.12, $P \leq 0.002$ for the first axis, λ =

1.008, $F = 8.25$, $P \leq 0.002$ for the overall ordination). The first axis was a topographic gradient that included the effects of elevation, slope, and RdNBR (Table 6-2). The second axis was a gradient in rainfall distinguishing the early versus late wet season (21.5% variation explained), and the third axis an aspect gradient (9.5% variation explained). Patterns of woody species composition varied in a similar fashion as those of herbaceous species. The natural regeneration, mesic blackbrush, and pinyon-juniper plots were distinctly separated along the topographic gradient of the first CCA axis (Figure 6-3A), as were the plots representing the three sampling years distinctly separated along the wet season rainfall gradient represented by the second CCA axis (Figure 6-3B). Individual woody species and guilds (shrubs, trees) also varied significantly along these same two CCA axes, with temporal shifts in spatial patterns occurring within each community in response to the timing and amount of rainfall (Figure 6-4).

Table 6-2. Correlation matrix and ordination statistics from a Canonical Correspondence Analysis of woody species composition for four vegetation types burned in the southern Nevada Complex fires of 2005. λ is the eigenvalue (variance) for each axis, and RdNBR is an index of fire severity.

Variable	Axis 1	Axis 2	Axis 3
Elevation	-0.978	0.001	-0.099
Slope	-0.615	-0.002	0.434
Aspect	-0.234	0.007	0.619
RdNBR	-0.587	-0.062	-0.414
Late dry season precipitation	-0.253	0.157	-0.170
Early wet season precipitation	-0.102	0.840	-0.232
Late wet season precipitation	-0.126	-0.913	-0.030
Early dry season precipitation	-0.105	0.457	-0.242
	Axis 1	**Axis 2**	**Axis 3**
λ :	0.464	0.217	0.095
Species-environment correlations	0.855	0.715	0.513
Cumulative percentage variance			
species data	3.5	5.1	5.8
species-environment relation	46.1	67.6	77.1

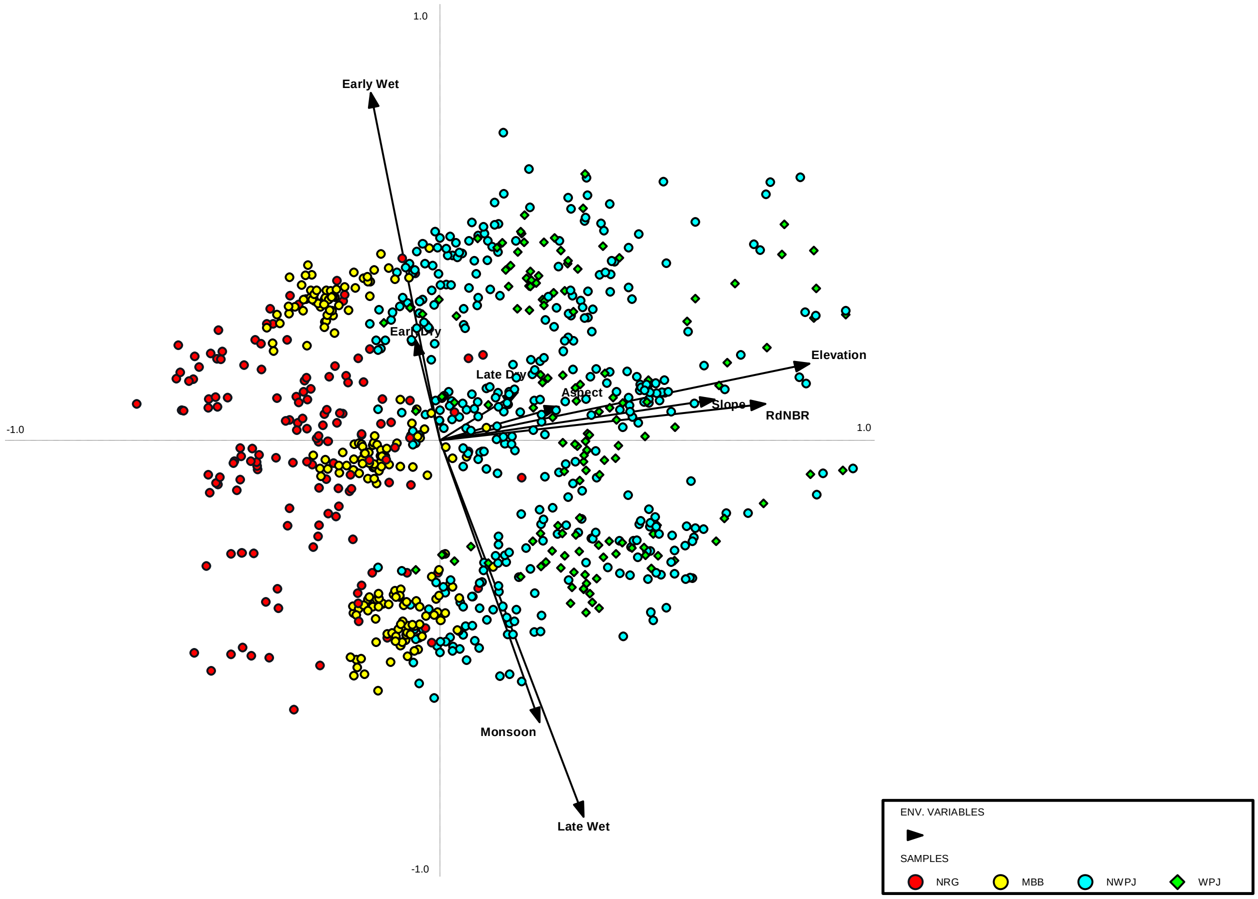


Figure 6-1A. Canonical Correspondence Analysis biplots of changes in herbaceous species composition for plots in different vegetation types. All plots were burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity.

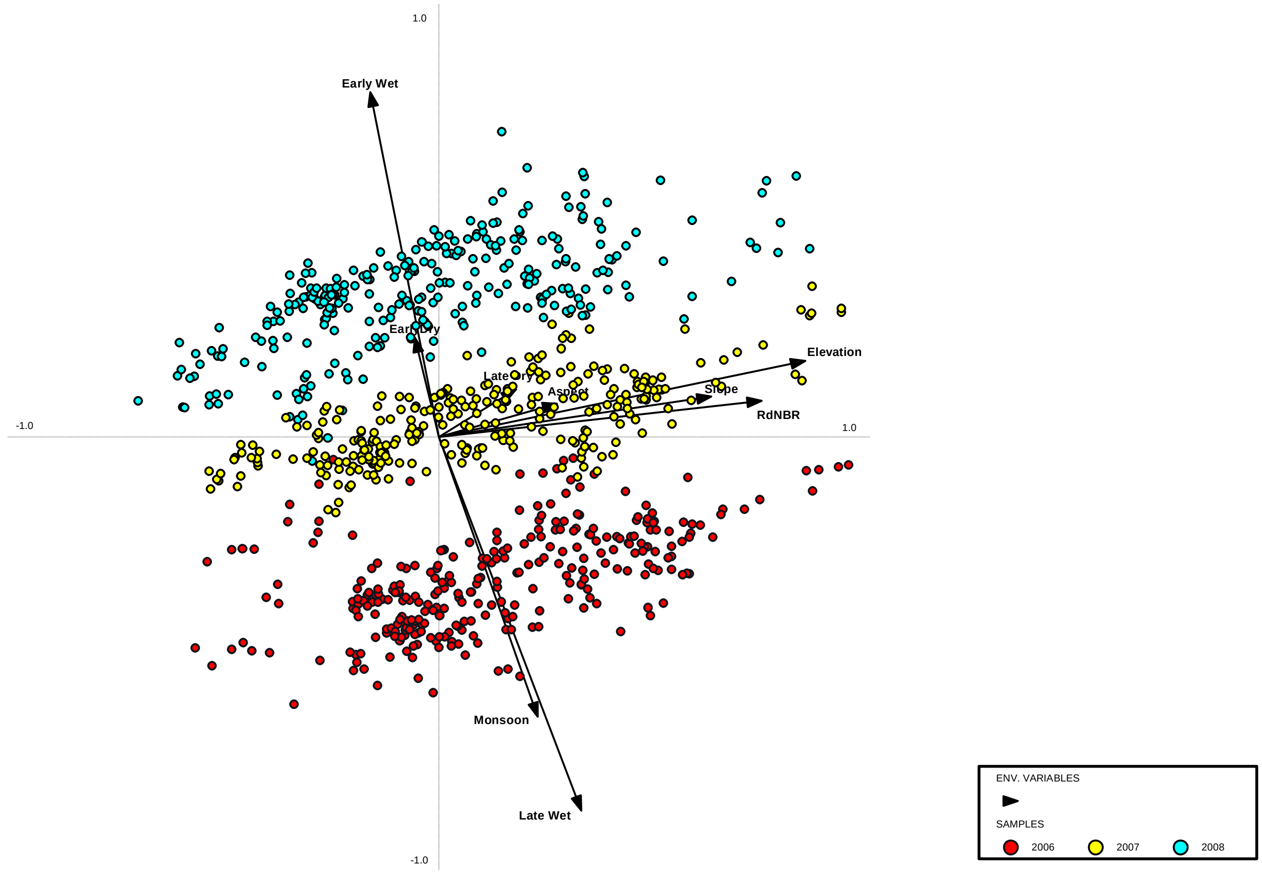


Figure 6-1B. Canonical Correspondence Analysis biplots of changes in herbaceous species composition for plots in years with different rainfall patterns (bottom panel). All plots were burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity. 2006 was the first year post-burn.

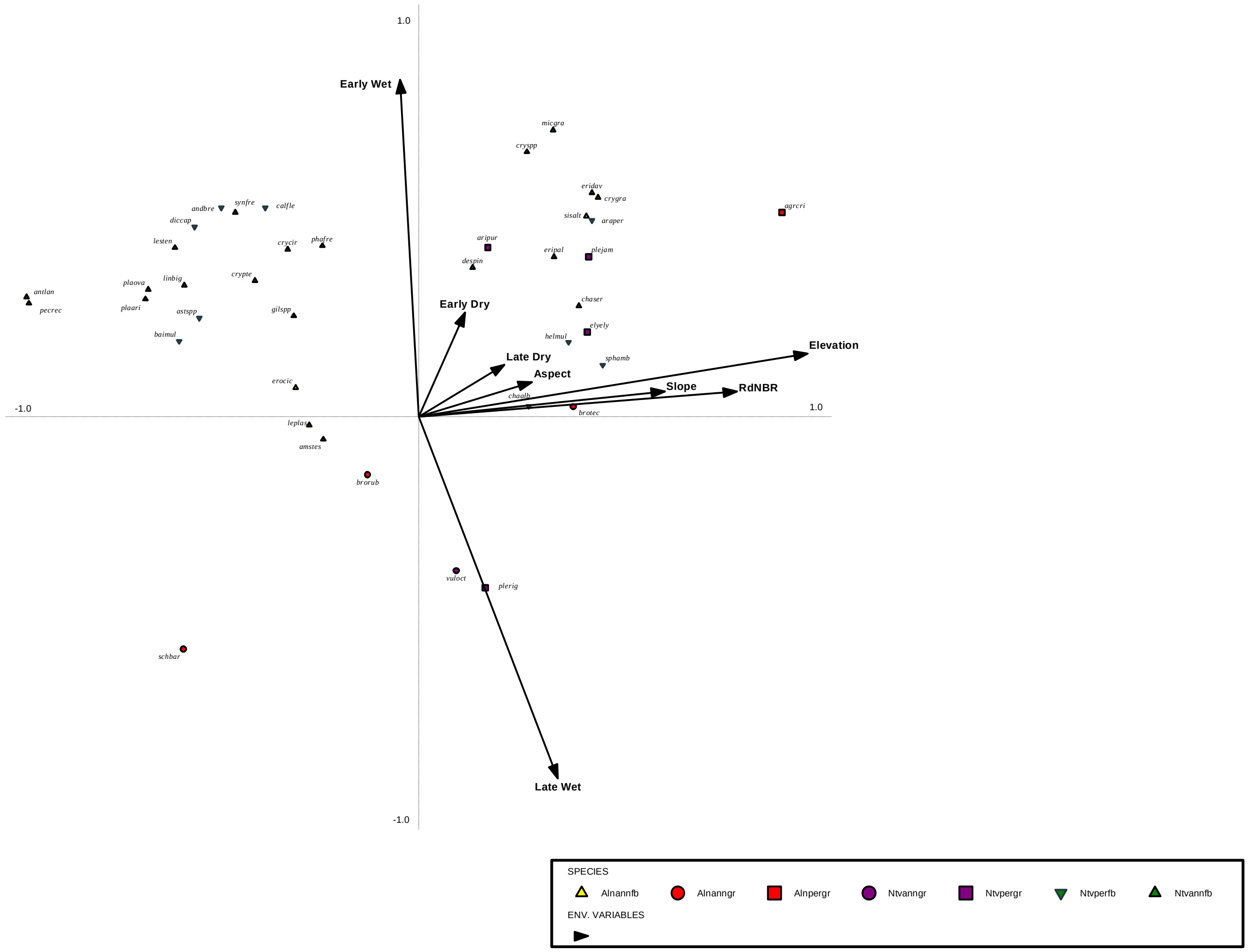


Figure 6-2. Canonical Correspondence Analysis biplot of herbaceous species composition in plots that burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity. Species acronyms are given in Appendix 6-3.

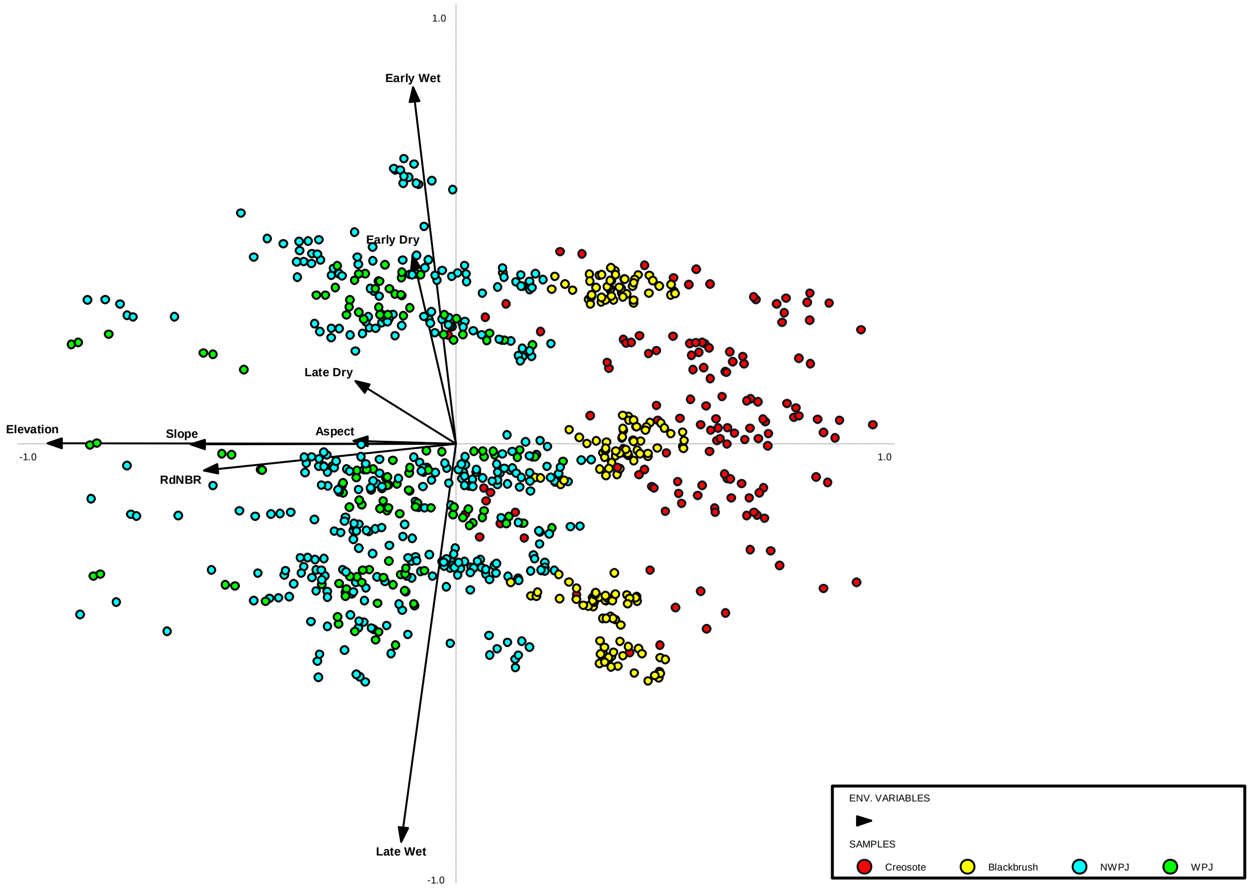


Figure 6-3A. Canonical Correspondence Analysis biplot of changes in woody species composition for plots in different vegetation types. All plots were burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity.

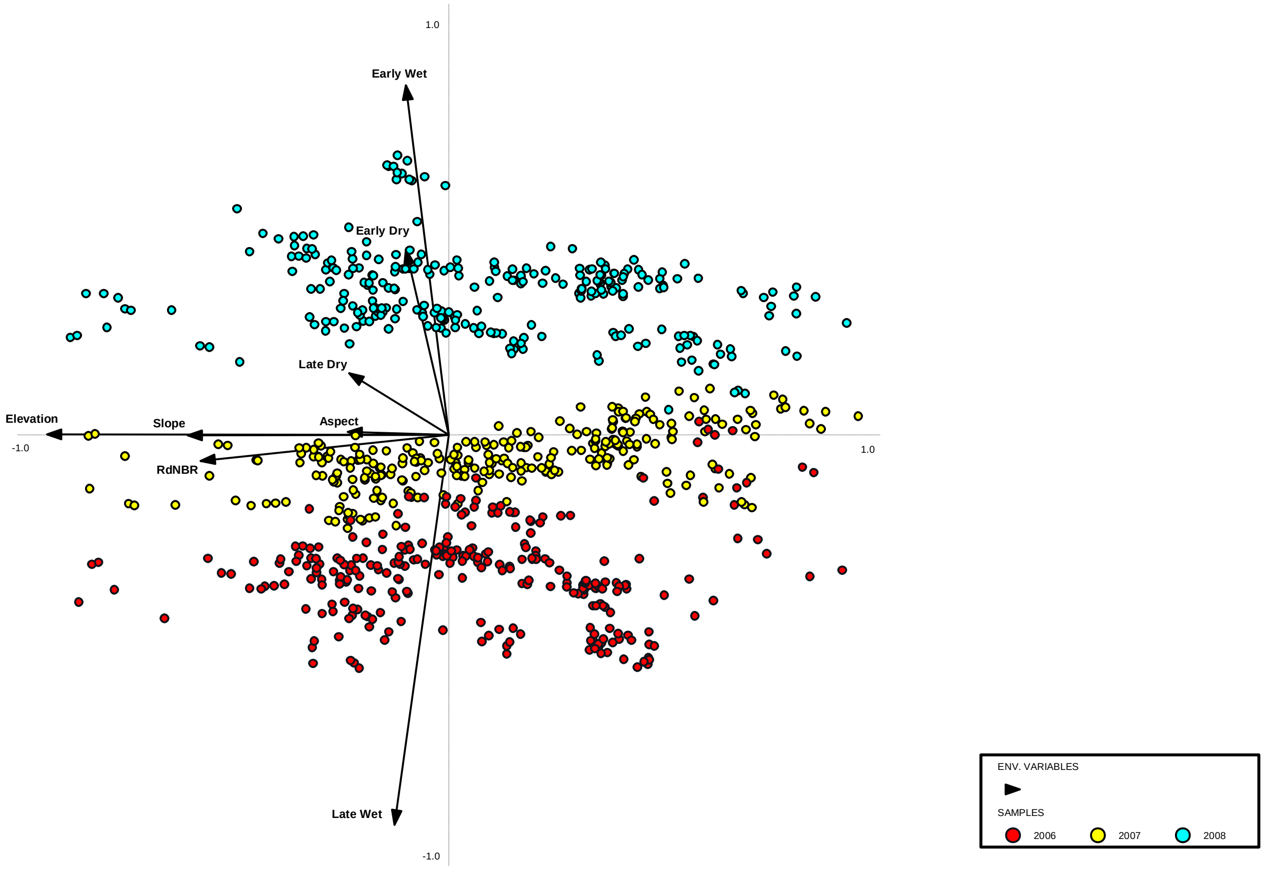


Figure 6-3B. Canonical Correspondence Analysis biplot of changes in woody species composition for plots in years with different rainfall patterns. All plots were burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity. 2006 was the first year post-burn.

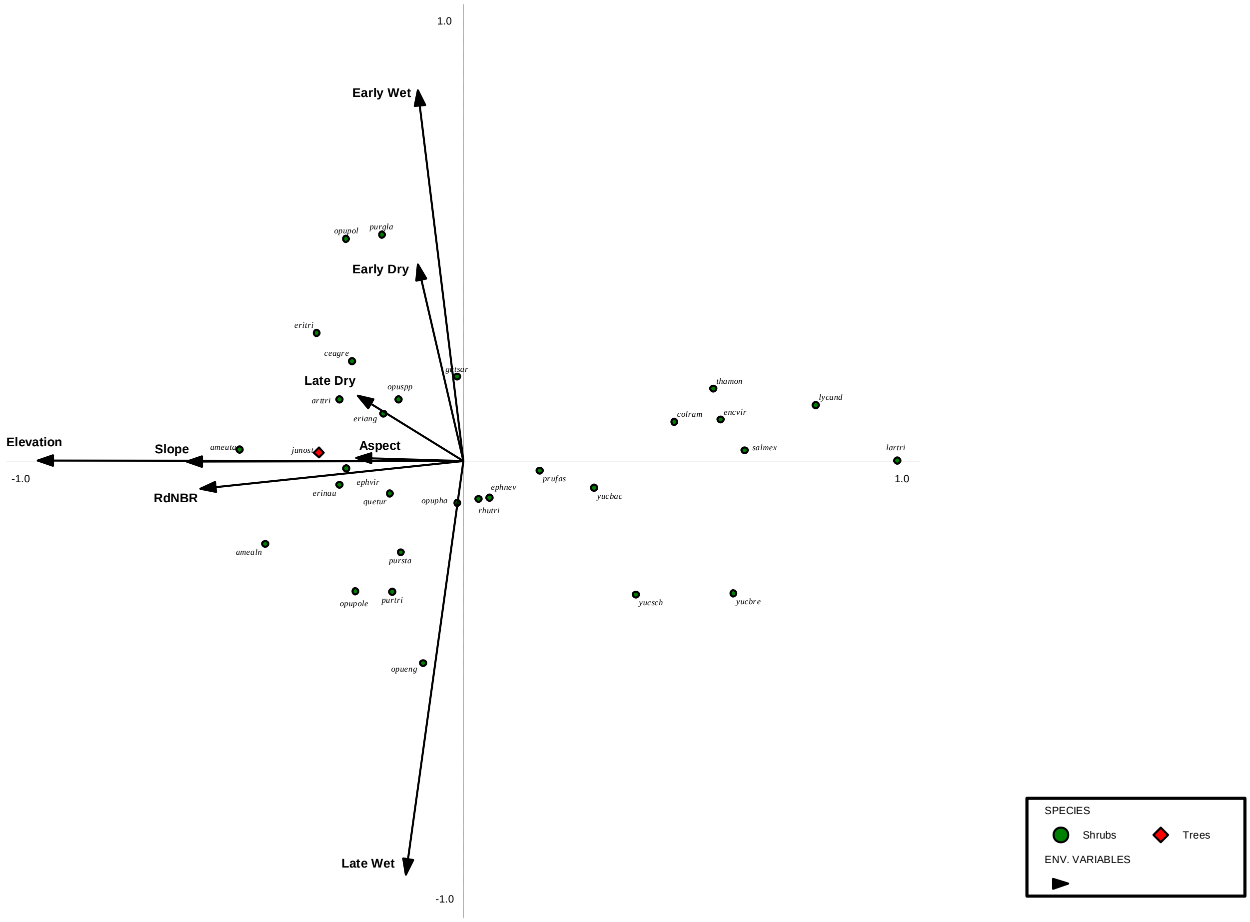


Figure 6-4. Canonical Correspondence Analysis biplot of woody species composition in plots that burned in the Southern Nevada Complex fires of 2005. RdNBR is an index of fire severity. Species acronyms are given in Appendix 6-3.

Total Herbaceous Species Diversity and Density

The species diversity indices N_0, N_1, N_2, and $E_{1/d}$ varied among vegetation types across the three years of the study (Figure 6-5). With the exception of $E_{1/d}$, the most complex models had the greatest support for each of the response variables (Appendix 6-1); the model with the greatest support for $E_{1/d}$ was the second most complex model, lacking only a quadratic time component (Appendix 6-1). The most complex model for $E_{1/d}$ had the second greatest level of support, and the relative AIC weights were only marginally different between it and the best supported model (Appendix 6-1).

Variation in the four diversity indices was characterized by non-linear and linear interactions between vegetation type and time (Appendix 6-2). N_0 declined between 2006 and 2007 in natural regeneration and mesic blackbrush communities, but did not change significantly in the two pinyon-juniper communities between these two years; it then increased in 2008 to its highest level in the study in all four vegetation types (Figure 6-5). N_1 and N_2 also declined between 2006 and 2007, but in 2008 the values for each index had increased in all of the vegetation types except mesic blackbrush (Figure 6-5). $E_{1/d}$ declined linearly in mesic blackbrush and wilderness pinyon-juniper communities, but increased linearly in natural regeneration and non-wilderness pinyon-juniper communities (Figure 6-5). Variation in these patterns among the four fires was $< 3\%$ (Appendix 6-2).

The patterns of change in overall density (i.e. pooled across all guilds) also varied among vegetation types across the three years of the study (Figure 6-6). Mesic blackbrush displayed a linear increase, wilderness and non-wilderness pinyon-juniper showed no change, and the natural regeneration plots responded with variable density among the three years.

Native Herbaceous Guild Diversity and Density

The absolute and relative species richness and density of the four native herbaceous guilds had different temporal patterns among the vegetation types (Appendix 6-3). The models with the greatest support for native annual and perennial forbs and native annual grasses included the linear and/or quadratic effects of time and their interaction with vegetation type (Appendix 6-3 and Appendix 6-4). However, the models with the greatest support for native perennial grasses did not include a temporal component, but only differences among vegetation types (Appendix 6-3 and Appendix 6-4). These patterns were consistent across the four fires, with less than 5% of the variation attributed to differences among them (Appendix 6-4).

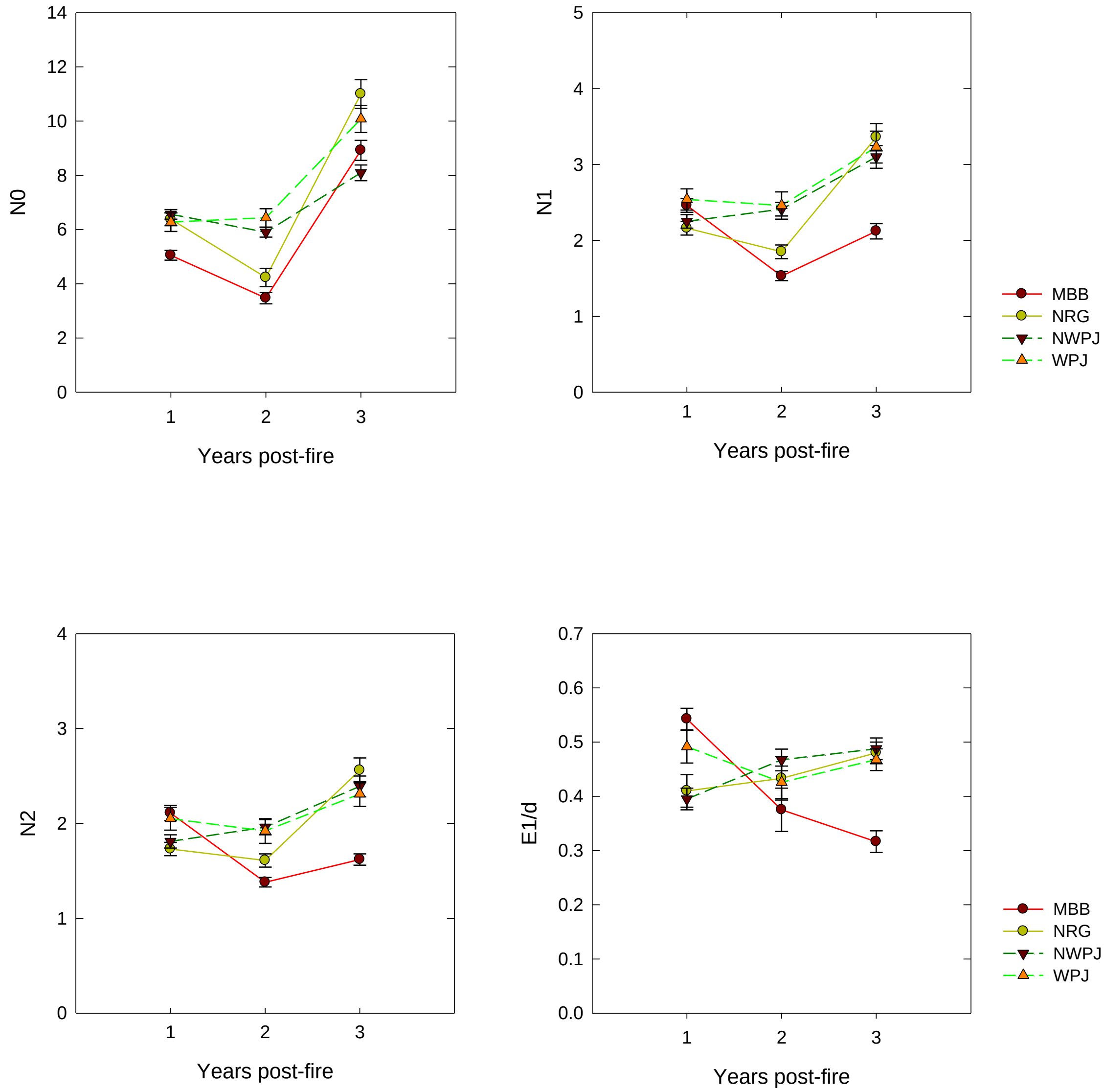


Figure 6-5. Variation in four indices of herbaceous species diversity in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). N_0 is overall species richness, N_1 is the exponentiation of Shannon's index (H'), N_2 is the reciprocal of Simpson's index of concentration (d), and $E_{1/d}$ is Simpson's index of evenness. The first post-fire year was 2006.

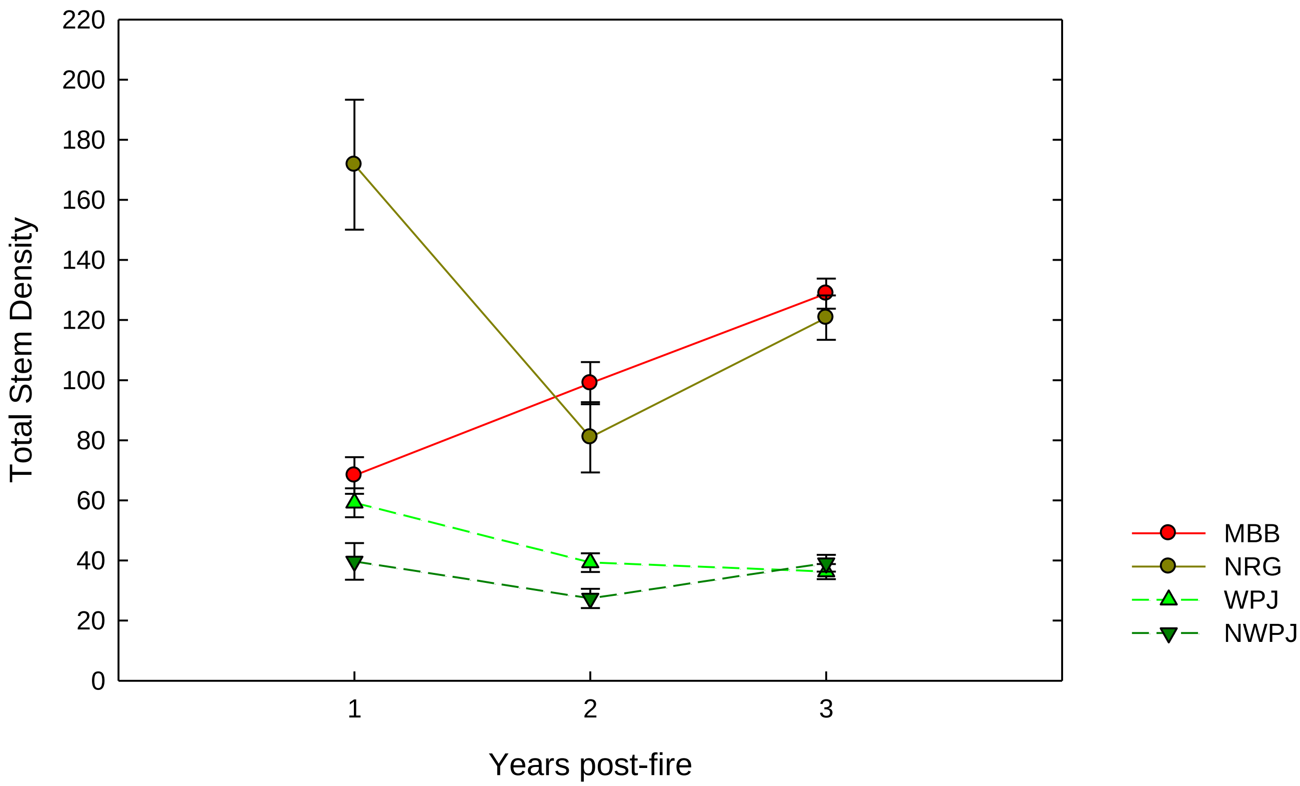


Figure 6-6. Variation in total herbaceous stem density (stems per m^2) in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The first post-fire year was 2006.

The mean number of native annual forb species increased in all vegetation types between 2006 and 2008, while the mean number of native perennial forb species increased in natural regeneration and mesic blackbrush vegetation types (Table 6-3). The mean number of native annual and perennial grass species did not vary significantly in natural regeneration and mesic blackbrush, but they declined in both pinyon-juniper vegetation types. Native annual forbs comprised < 30% of the species in 2006 and 2007, but by 2008 they comprised 35% to almost 50% in all vegetation types (Figure 6-7). The mean percentage of native perennial forb species in the pinyon-juniper communities decreased from 40% - 45% in 2006 and 2007 to approximately 25% in 2008 (Figure 6-7). Native grasses comprised < 10% of the species in all vegetation types (Figure 6-7).

Table 6-3. The mean number of species ($m^{-2} \pm SE$) in four native herbaceous guilds in four vegetation types that burned in the Southern Nevada Complex fires of 2005. Vegetation in the "Natural Regeneration" category was Creosote/Thermic blackbrush. PJ = pinyon-juniper.

Vegetation	**2006**	**2007**	**2008**	**Mean**
Native Annual Forbs				
Mesic Blackbrush	0.89 (0.11)	0.33 (0.08)	3.65 (0.25)	1.57 (0.13)
Natural Regeneration	1.81 (0.17)	0.88 (0.16)	5.55 (0.43)	2.96 (0.25)
Non-wilderness PJ	1.46 (0.09)	1.06 (0.09)	2.77 (0.20)	1.73 (0.08)
Wilderness PJ	1.20 (0.18)	1.28 (0.18)	4.23 (0.33)	2.24 (0.18)
Mean	1.32 (0.06)	0.91 (0.06)	3.78 (0.16)	1.99 (0.07)
Native Perennial Forbs				
Mesic Blackbrush	0.37 (0.08)	0.52 (0.09)	2.52 (0.19)	1.08 (0.10)
Natural Regeneration	1.03 (0.20)	0.87 (0.14)	2.62 (0.18)	1.60 (0.12)
Non-wilderness PJ	2.27 (0.11)	2.26 (0.11)	2.28 (0.12)	2.27 (0.07)
Wilderness PJ	2.35 (0.22)	2.66 (0.22)	2.71 (0.23)	2.57 (0.13)
Mean	1.65 (0.09)	1.71 (0.09)	2.47 (0.08)	1.94 (0.05)
Native Annual Grass				
Mesic Blackbrush	0.53 (0.06)	0.08 (0.04)	0.35 (0.07)	0.34 (0.04)
Natural Regeneration	0.51 (0.10)	0.06 (0.03)	0.17 (0.05)	0.22 (0.04)
Non-wilderness PJ	0.39 (0.05)	0.07 (0.02)	0.03 (0.02)	0.17 (0.02)
Wilderness PJ	0.33 (0.07)	0.17 (0.06)	0.15 (0.05)	0.22 (0.03)
Mean	0.43 (0.03)	0.09 (0.02)	0.15 (0.02)	0.22 (0.01)
Native Perennial Grass				
Mesic Blackbrush	0.11 (0.04)	0.08 (0.04)	0.23 (0.06)	0.14 (0.03)
Natural Regeneration	0.08 (0.05)	0.19 (0.06)	0.31 (0.07)	0.21 (0.04)
Non-wilderness PJ	0.38 (0.05)	0.44 (0.06)	0.35 (0.05)	0.39 (0.03)
Wilderness PJ	0.37 (0.10)	0.43 (0.10)	0.48 (0.10)	0.42 (0.06)
Mean	0.27 (0.03)	0.32 (0.04)	0.34 (0.03)	0.31 (0.02)

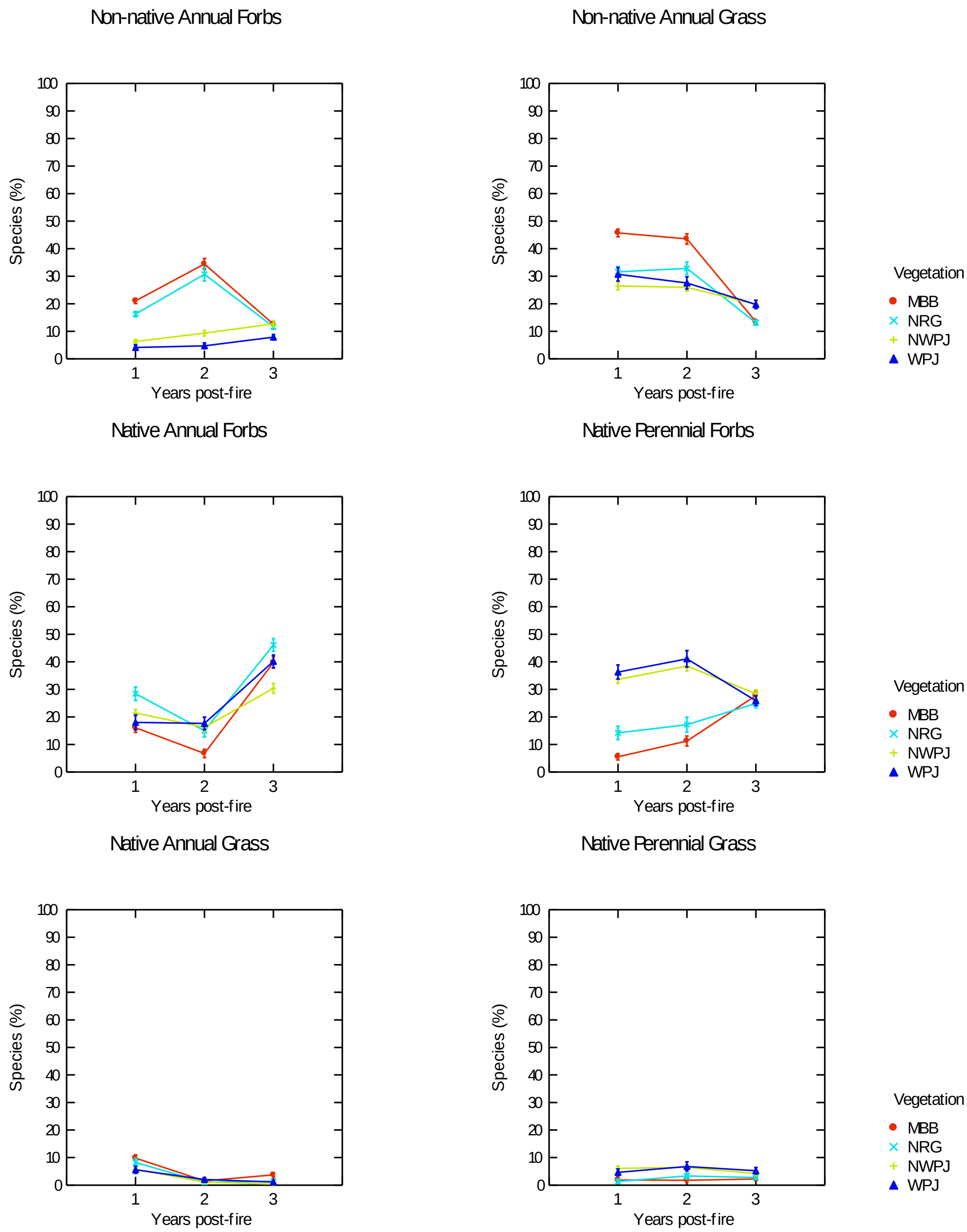


Figure 6-7. The percentage of species in six herbaceous plant guilds in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The first post-fire year was 2006.

The mean number of stems in four native herbaceous guilds is presented in Table 6-4. Density of native annual forbs increased between 2007 and 2008 in all vegetation types. Density of native perennial forbs decreased in natural regeneration and mesic blackbrush between 2006 and 2007, but in 2008 it increased to the same level as in 2006. Density of native annual grasses was characterized by high levels of variability so patterns were not clear, but a downward trend appeared to occur in all vegetation types except mesic blackbrush. Density of native perennial grasses declined in non-wilderness pinyon-juniper. Native forbs comprised < 25% of the herbaceous density and native grass < 10% in all vegetation types in all years (Figure 6-8).

Table 6-4. The mean number of individuals ($m^{-2} \pm SE$) in four native herbaceous guilds in four vegetation types that burned in the Southern Nevada Complex fires of 2005. Vegetation in the "Natural Regeneration" category was Creosote/Thermic blackbrush. PJ = pinyon-juniper.

Vegetation	2006	2007	2008	Mean
Native Annual Forbs				
Mesic Blackbrush	0.36 (0.10)	0.12 (0.04)	6.74 (0.99)	2.25 (0.37)
Natural Regeneration	3.08 (0.73)	1.22 (0.66)	16.90 (2.28)	7.88 (1.12)
Non-wilderness PJ	2.34 (0.78)	1.02 (0.34)	2.59 (0.35)	1.97 (0.32)
Wilderness PJ	1.09 (0.30)	0.90 (0.32)	4.55 (0.76)	2.18 (0.32)
Mean	1.72 (0.37)	0.85 (0.20)	6.75 (0.63)	3.08 (0.27)
Native Perennial Forbs				
Mesic Blackbrush	0.27 (0.09)	0.17 (0.04)	3.63 (0.56)	1.27 (0.21)
Natural Regeneration	1.40 (0.51)	0.36 (0.09)	3.11 (0.45)	1.71 (0.24)
Non-wilderness PJ	1.48 (0.15)	3.85 (0.49)	1.83 (0.20)	2.38 (0.19)
Wilderness PJ	1.87 (0.41)	3.19 (0.93)	1.68 (0.23)	2.24 (0.34)
Mean	1.23 (0.12)	2.33 (0.28)	2.45 (0.18)	1.99 (0.12)
Native Annual Grass				
Mesic Blackbrush	10.10 (3.76)	0.03 (0.01)	0.38 (0.11)	4.01 (1.48)
Natural Regeneration	6.12 (2.91)	3.35 (2.34)	1.33 (0.95)	3.25 (1.17)
Non-wilderness PJ	2.35 (0.65)	0.21 (0.09)	0.03 (0.02)	0.92 (0.24)
Wilderness PJ	1.86 (1.10)	0.25 (0.14)	0.04 (0.02)	0.73 (0.38)
Mean	4.70 (1.09)	0.75 (0.43)	0.38 (0.20)	1.98 (0.41)
Native Perennial Grass				
Mesic Blackbrush	0.06 (0.03)	0.05 (0.03)	0.06 (0.02)	0.06 (0.02)
Natural Regeneration	0.07 (0.05)	0.11 (0.06)	0.21 (0.11)	0.14 (0.05)
Non-wilderness PJ	0.45 (0.19)	0.28 (0.07)	0.12 (0.03)	0.29 (0.07)
Wilderness PJ	0.70 (0.40)	0.26 (0.16)	0.29 (0.12)	0.42 (0.15)
Mean	0.35 (0.11)	0.20 (0.04)	0.16 (0.03)	0.24 (0.04)

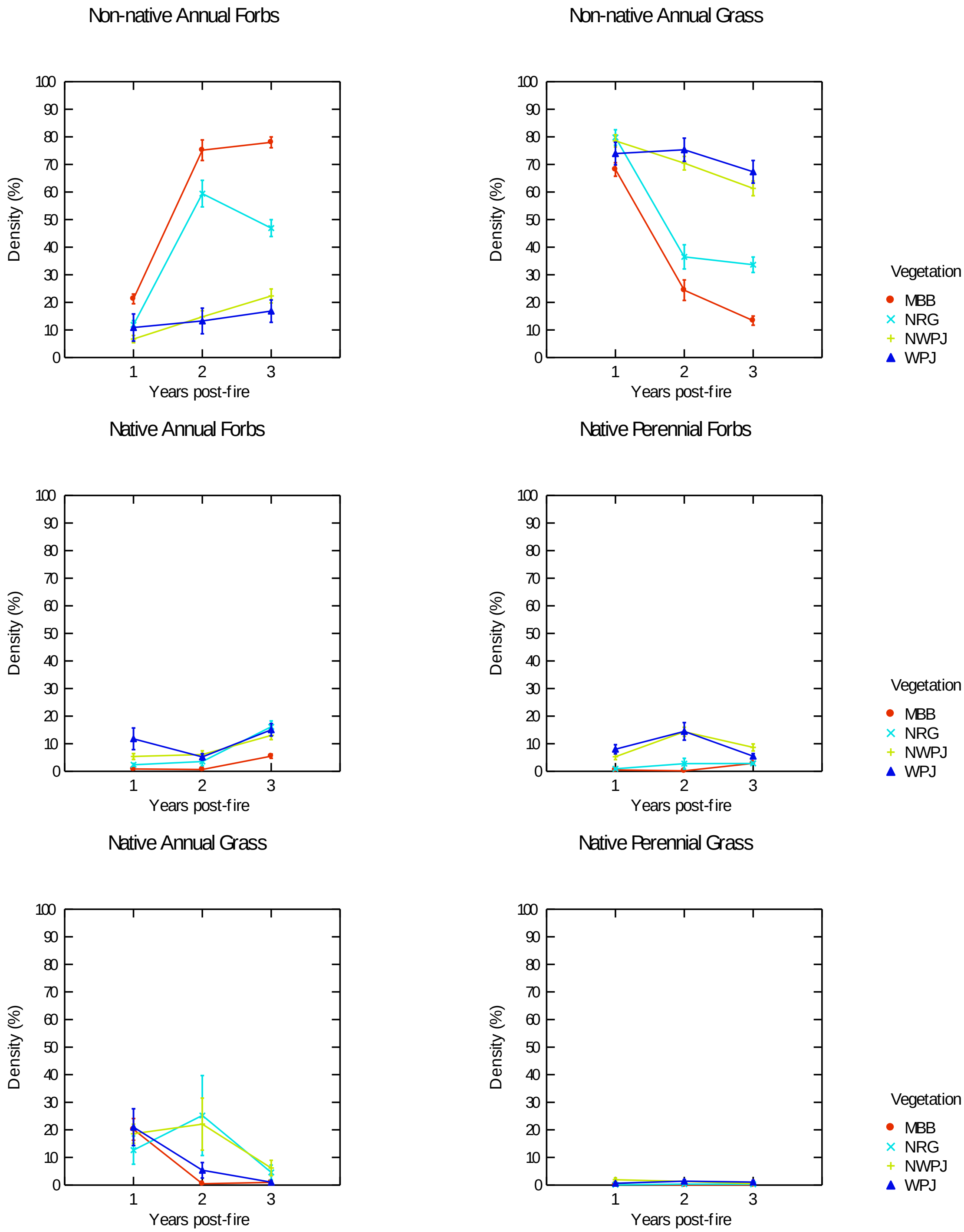


Figure 6-8. The percentage of stems in six herbaceous plant guilds in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The first post-fire year was 2006.

There was no significant relationship between RdNBR and density or species richness of any of the native guilds ($P \geq 0.2182$). Species richness of native annual forbs and density and species richness of native annual grasses had strong positive relationships with precipitation ($P \leq 0.0001$).

Woody Species Diversity and Density

All indices of woody diversity were greater in non-wilderness and wilderness pinyon-juniper communities than natural regeneration and mesic blackbrush communities in all years (Figure 6-9). There was no evidence of any temporal variation in these patterns (Appendices 6-5 and 6-6).

Density (resprouts and mature individuals) of shrubs and trees was higher in pinyon-juniper communities than natural regeneration and mesic blackbrush communities in all years (Figure 6-10). There was no evidence of any temporal trends in these patterns (Appendices 6-5 and 6-6).

Seedling density declined in all vegetation types between 2006 and 2008 (Figure 6-10), but the decline was linear in natural regeneration and mesic blackbrush communities and non-linear in non-wilderness and wilderness pinyon-juniper communities (Appendices 6-5 and 6-6).

There was no relationship between woody density and precipitation or RdNBR ($P \geq 0.4087$). Density of woody seedlings had a strong positive relationship with precipitation ($P \leq 0.0001$).

Non-Native Herbaceous Guild Density

The absolute and relative proportion of species and density for non-native grasses and forbs not only varied among vegetation types, but also had different, often non-linear temporal patterns among the vegetation types (Appendix 6-3). Moreover, the patterns were very consistent among the Delamar, Duzak, Halfway, and Meadow Valley fires, as variation among the four fires accounted for less than 5% of the variation in the response variables (Appendix 6-4). The models with the greatest support included the linear and quadratic effects of time and their interaction with vegetation type; there was virtually no support for the other models ($\Delta AIC \geq 10.83$). Collectively, non-native grasses and forbs comprised 81% to 94% of herbaceous density in natural regeneration plots, 88% to 99% in mesic blackbrush, and 77% to 82% in wilderness and non-wilderness pinyon-juniper plots.

Despite their dominance of the herbaceous vegetation in all of the vegetation types, non-native grasses and forbs showed very different spatial and temporal patterns. Non-native grasses comprised approximately half of the species in mesic blackbrush plots in 2006 and

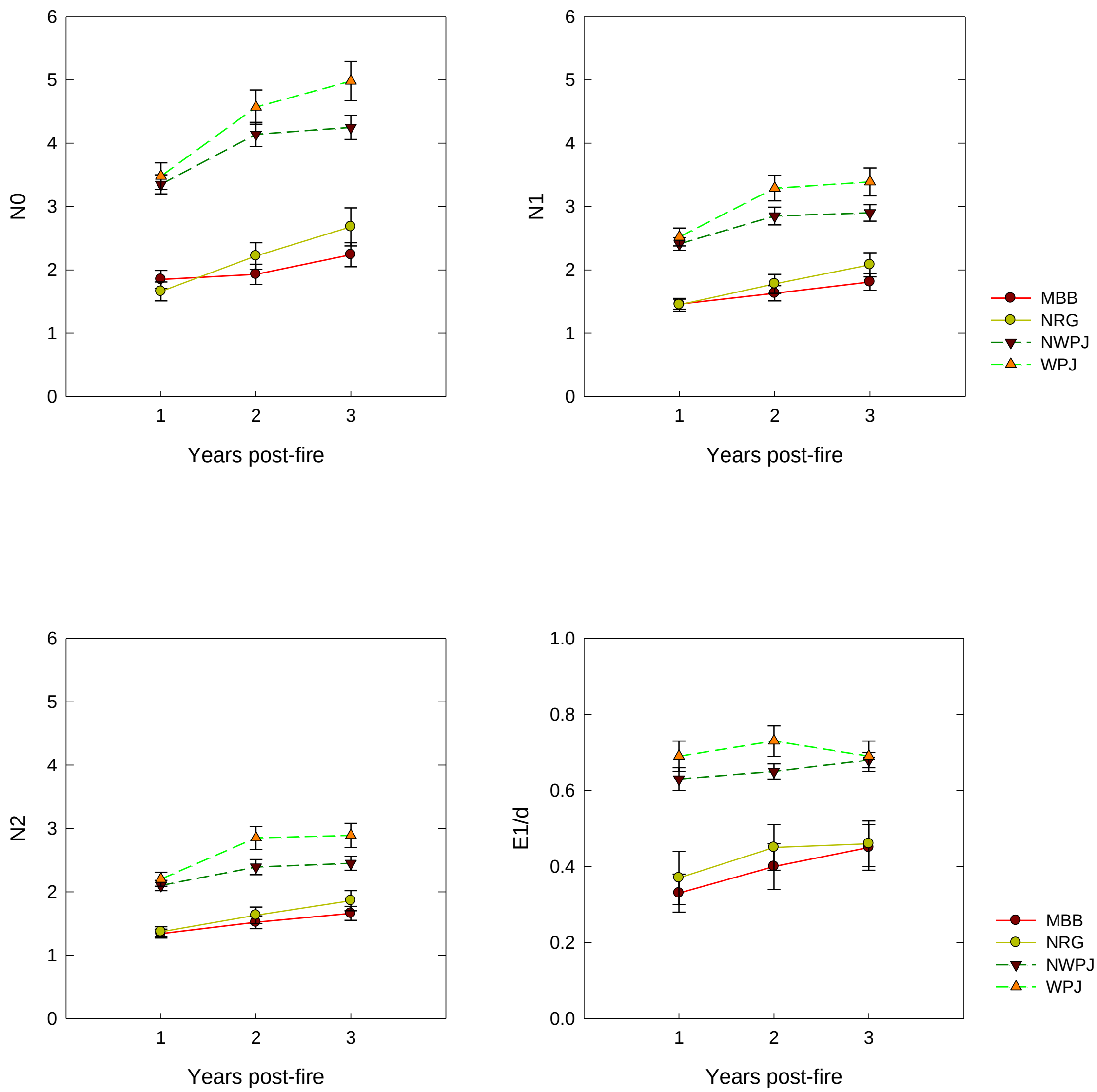
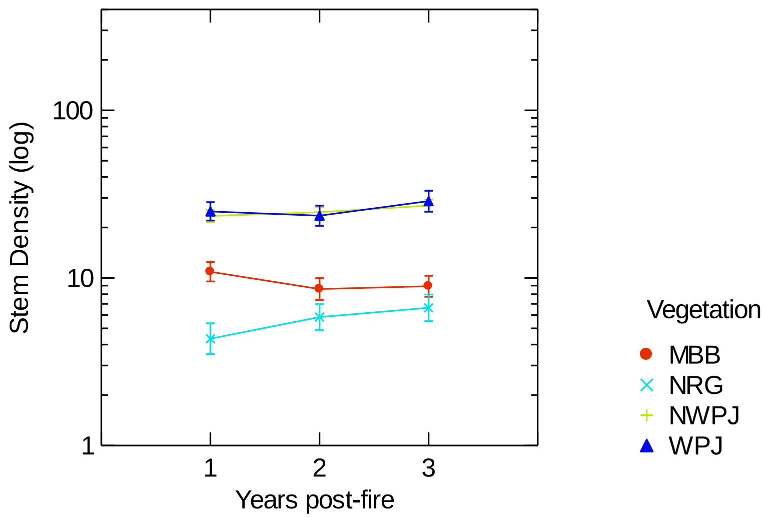


Figure 6-9. Variation in four indices of woody species diversity in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). N_0 is overall species richness, N_1 is the exponentiation of Shannon's index (H'), N_2 is the reciprocal of Simpson's index of concentration (d), and $E_{1/d}$ is Simpson's index of evenness. The first post-fire year was 2006.



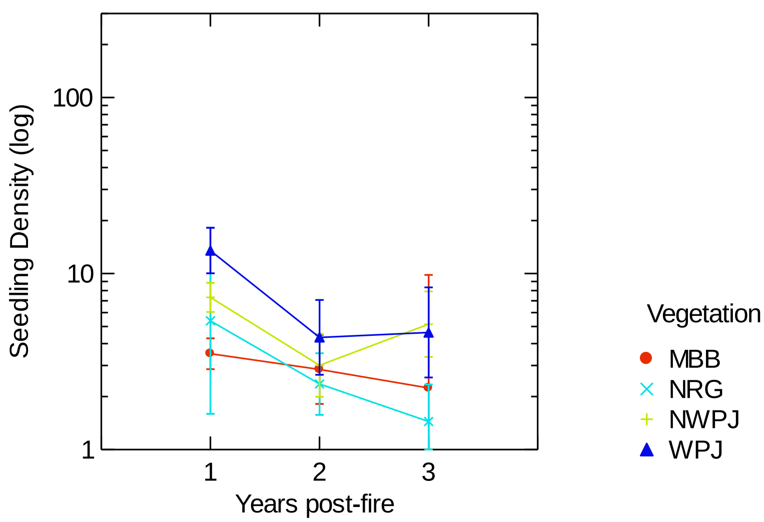


Figure 6-10. The density of woody stems (per 150 m^2) and seedlings (per m^2) in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The first post-fire year was 2006.

2007, but then dropped to approximately 15% in 2008 (Figure 6-7). A similar pattern occurred in natural regeneration and pinyon-juniper plots, except that non-native grasses comprised only 25% to 35% of the species in 2006 and 2007. The lower percentage of species in the four vegetation types in 2008 was not due to a decrease in the presence of non-native grass species; species numbers remained relatively constant in the four types (6 in

natural regeneration, 5-6 in blackbrush, 7-11 in NWPJ, and 4-8 in WPJ). Rather, the change was associated with an increase in native species, especially annual forbs (Figure 6-7).

The percentage of non-native forb species in natural regeneration and mesic blackbrush plots peaked at 30% to 35% in 2007 (Figure 6-7). The percentage of non-native forb species in wilderness and non-wilderness pinyon-juniper plots increased linearly from 2006 to 2008, but they comprised less than 10% of the species in this vegetation type.

The relative density of non-native grasses decreased linearly by almost 60% in mesic blackbrush plots (Figure 6-8). There was a similar pattern in natural regeneration plots except the decline was more non-linear and not as severe; relative density declined from 80% to 40% between 2006 and 2007, but was similar between 2007 and 2008. Relative density of non-native grasses also declined in wilderness and non-wilderness pinyon-juniper plots, but the decline was far less severe and they continued to comprise > 60% of the herbaceous density in these vegetation types (Figure 6-8).

The percentage of non-native forb stems increased in all vegetation types (Figure 6-8). The pattern of increase in natural regeneration and mesic blackbrush plots was nonlinear, while that in wilderness and non-wilderness pinyon-juniper plots was linear. By 2008, non-native forbs comprised 80% of the stems in mesic blackbrush plots, 50% to 60% in natural regeneration plots, and 12% to 18% in wilderness and non-wilderness pinyon-juniper plots.

A total of 19 non-native herbaceous species were recorded in the samples (Table 6-5). There were no non-native woody species recorded in the samples. Six of the eight grass species were annuals, as were eight of the eleven forb species. Density was dominated by cheatgrass, red brome, and *Erodium*; collectively these three species accounted for 72% of non-native density and 84% of total herbaceous density. Mean density of cheatgrass, red brome, and *Erodium* was 24.4, 28.1, and 33.5 individuals m^{-2} respectively, but maximum density of these three species could be more than an order of magnitude greater (Table 6-5). *Schismus* spp. accounted for an additional 17% of non-native density, but they were much more restricted in distribution than cheatgrass, red brome, and *Erodium*. Cheatgrass, red brome, and *Erodium* occurred in 65% to 73% of the plots; collectively they occurred in 98% of the plots. *Schismus* spp. occurred in less than 6% of the plots, primarily located in the lower elevation natural regeneration plots.

The mean densities of cheatgrass, red brome, and *Erodium* in the four burned areas are presented in Table 6-6. There was no distinct spatial pattern in abundance among the burned areas, which is expected because they are geographic units and not meaningful ecological strata. In contrast, spatial patterns were apparent when the mean densities of the three species were broken out by vegetation type (Table 6-7). Although all three species

occurred in all of the vegetation types, red brome and *Erodium* were much more abundant in the natural regeneration plots (creosotebush and thermic blackbrush) and to a lesser degree in the mesic blackbrush plots, and cheatgrass was most abundant in the pinyon-juniper plots. Whether broken out by vegetation type or burned area, all three species showed clear temporal patterns in abundance. Red brome and cheatgrass decreased in abundance between 2006 and 2007, then remained relatively low in 2008 (Tables 6-6 and 6-7). In contrast, *Erodium* had a linear increase in abundance between 2006 and 2008.

Table 6-5. Non-native plant species recorded in vegetation sampling plots in four of the Southern Nevada Complex fires of 2005. The fires were Delamar, Duzak, Halfway, and Meadow Valley. There were 355 plots sampled from 1-3 times between 2006 and 2008 for a total of 863 sample events. Values for the total number of plots [Plots (N)] and the percent of plots [Plots (%)] were based on the 863 sample events. Stem density is the number of stems per m^2.

Species	Life History	Life Form	Plots (N)	Plots (%)	Stem Density (Mean)	Stem Density (Maximum)
Agropyron cristatum	Perennial	Grass	48	5.6	0.7	6.8
Agropyron fragile	Perennial	Grass	2	0.2	0.4	0.6
Brassica tournefortii	Annual	Forb	2	0.2	2.8	4.2
Bromus arvensis	Annual	Grass	5	0.6	2.4	4.7
Bromus rubens	Annual	Grass	633	73.3	28.1	467.5
Bromus tectorum	Annual	Grass	619	71.7	24.4	395.0
Bromus trinii	Annual	Grass	6	0.7	2.8	10.0
Convolvulus arvensis	Perennial	Forb	4	0.5	0.2	0.5
Erodium cicutarium	Annual	Forb	557	64.5	33.5	239.0
Halogeton glomeratus	Annual	Forb	2	0.2	0.2	0.2
Lactuca serriola	Annual	Forb	11	1.3	0.7	3.2
Lotus corniculatus	Perennial	Forb	2	0.2	0.4	0.6
Plantago lanceolata	Annual	Forb	2	0.2	0.1	0.1
Polygonum argyrocoleon	Annual	Forb	2	0.2	0.1	0.1
Salsola tragus	Annual	Forb	6	0.7	0.4	1.2
Sanguisorba minor	Perennial	Forb	6	0.7	0.2	0.7
Schismus arabicus	Annual	Grass	3	0.3	11.0	21.4
Schismus barbatus	Annual	Grass	53	6.1	9.7	130.2
Sisymbrium altissimum	Annual	Forb	66	7.6	2.2	38.5

Table 6-6. Mean density (individuals m^{-2} ± SE) of three non-native herbaceous species (*Bromus rubens*, *Bromus tectorum*, and *Erodium cicutarium*) in four areas that burned in the Southern Nevada Complex fires of 2005. MV is the Meadow Valley fire.

Species			**Year**					
B. rubens	**2006**		**2007**		**2008**		**Mean**	
Delamar	14.9	(3.8)	17.8	(6.0)	11.4	(3.8)	14.8	(2.7)
Duzak	31.7	(3.0)	11.0	(1.4)	12.6	(1.6)	18.8	(1.3)
Halfway	150.9	(54.1)	16.8	(7.2)	35.6	(7.7)	53.3	(14.3)
MV	44.7	(7.5)	15.7	(6.2)	11.8	(3.5)	24.3	(3.7)
Mean	34.0	(3.2)	13.6	(1.9)	13.7	(1.4)	20.6	(1.4)
B. tectorum								
Delamar	32.4	(5.6)	30.7	(3.7)	27.8	(4.2)	30.4	(2.7)
Duzak	20.3	(3.5)	9.4	(1.8)	11.4	(1.6)	13.9	(1.5)
Halfway	25.2	(9.1)	3.0	(2.0)	14.0	(6.3)	12.1	(3.4)
MV	14.7	(3.0)	16.9	(3.6)	10.0	(2.4)	13.9	(1.8)
Mean	22.4	(2.5)	15.0	(1.5)	14.9	(1.4)	17.5	(1.1)
E. cicutarium								
Delamar	1.2	(0.4)	6.7	(2.3)	12.5	(3.6)	6.6	(1.4)
Duzak	6.7	(0.6)	35.5	(4.3)	47.2	(4.0)	29.2	(2.1)
Halfway	40.0	(18.2)	28.2	(10.1)	47.5	(11.8)	38.3	(7.2)
MV	4.9	(1.1)	10.3	(2.9)	18.2	(5.0)	11.0	(2.0)
Mean	6.2	(0.8)	24.4	(2.7)	34.9	(2.8)	21.6	(1.3)

Table 6-7. Mean density (individuals m^{-2} ± SE) of three non-native herbaceous species (*Bromus rubens*, *Bromus tectorum*, and *Erodium cicutarium*) in four vegetation types that burned in the Southern Nevada Complex fires of 2005. CTB = Creosote/Thermic blackbrush (i.e. "natural regeneration" areas), NWPJ = non-wilderness pinyon-juniper, and WPJ = wilderness pinyon-juniper.

Species			**Year**					
B. rubens	**2006**		**2007**		**2008**		**Mean**	
CTB	108.7	(17.3)	32.1	(8.6)	38.1	(5.0)	53.7	(6.2)
Blackbrush	40.5	(4.2)	19.1	(2.9)	17.2	(2.6)	26.7	(2.2)
NWPJ	16.5	(2.3)	6.6	(1.4)	2.7	(0.5)	8.9	(1.0)
WPJ	15.8	(3.6)	5.2	(1.3)	6.2	(1.3)	9.1	(1.4)
Mean	34.0	(3.2)	13.6	(1.9)	13.7	(1.4)	20.6	(1.4)
B. tectorum								
CTB	18.4	(9.9)	5.9	(2.3)	6.5	(2.9)	9.3	(2.9)
Blackbrush	7.3	(1.4)	1.2	(0.4)	0.4	(0.2)	3.3	(0.6)
NWPJ	33.7	(4.3)	24.8	(2.8)	23.3	(2.6)	27.5	(2.0)
WPJ	17.2	(3.4)	16.4	(3.1)	23.0	(3.1)	18.9	(1.9)
Mean	22.4	(2.5)	15.0	(1.5)	14.9	(1.4)	17.5	(1.1)
E. cicutarium								
CTB	21.3	(5.3)	37.2	(5.6)	52.8	(5.0)	39.3	(3.2)
Blackbrush	9.4	(0.7)	78.1	(7.8)	100.3	(4.7)	58.5	(4.0)
NWPJ	2.2	(0.5)	2.2	(0.5)	5.3	(1.0)	3.1	(0.4)
WPJ	1.2	(0.5)	1.2	(0.5)	2.9	(0.8)	1.8	(0.3)
Mean	6.2	(0.8)	24.4	(2.7)	34.9	(2.8)	21.6	(1.3)

The species response curves highlighted the distinct patterns in abundance of cheatgrass, red brome, and *Erodium* along an elevation gradient (Figure 6-11). Unimodal response models for each species had overwhelmingly more support than linear models (ΔAIC $\geq$ 688.52). There was almost complete overlap in the distributions of the three species, but there was also separation of the peak responses among the species. Red brome and *Erodium* dominated lower elevation communities, with peak densities between 800 and 1200 meters, whereas cheatgrass dominated higher elevation communities, with a peak density at 1800 m. In addition, peak densities of red brome and *Erodium* were almost twice as high as cheatgrass, but the cumulative effect of the three species was consistently high across the elevation range.

Unimodal models of responses by red brome and *Erodium* to a gradient in RdNBR had overwhelmingly more support than linear models (ΔAIC $\geq$ 1656.51). Both species had almost identical response profiles, with their peak densities in the mid part of the RdNBR range (Figure 6-12). In contrast, there was no support for either a linear or unimodal response by cheatgrass to RdNBR (Figure 6-12). The constants only model had overwhelmingly more support than either linear or unimodal models (ΔAIC $\geq$ 114.15).

There was overwhelming support for unimodal models of red brome, *Erodium* and cheatgrass responses to precipitation (ΔAIC $\geq$ 71.36), but the patterns varied widely among the three species (Figure 6-13). The abundance of *Erodium* was greatest in drier conditions, with a near linear decrease in density as precipitation increased beyond average values. Density of red brome was greatest in conditions with average or somewhat greater precipitation, but then showed a rapid decline at higher levels of precipitation. The density of cheatgrass increased exponentially as precipitation increased beyond average values (Figure 6-13).

There was no evidence of a negative relationship between the density of cheatgrass, red brome, and *Erodium* (pooled) and density or richness of native annual herbaceous species or native perennial forbs. Species richness of native annual forbs and density of native perennial forbs actually had a positive relationship with density of cheatgrass, red brome, and *Erodium* ($P \leq 0.0436$). However, species richness of native perennial grass species had a negative relationship with density of cheatgrass, red brome, and *Erodium* ($P \leq 0.0001$).

Overall species richness (non-native and native herbaceous species; N_0) had a strong positive relationship with the density of cheatgrass, red brome, and *Erodium* ($P \leq 0.0005$). However, N_1, N_2, and $E_{1/d}$ had very strong negative relationships with density of cheatgrass, red brome, and *Erodium* ($P \leq 0.0001$).

Density of woody plant seedlings had a strong negative relationship with density of cheatgrass, red brome, and *Erodium*, and a strong positive relationship with precipitation ($P \leq 0.0001$). Density of woody seedlings was restricted to a relatively narrow range with high precipitation and low density of cheatgrass, red brome, and *Erodium* (Figure 6-14).

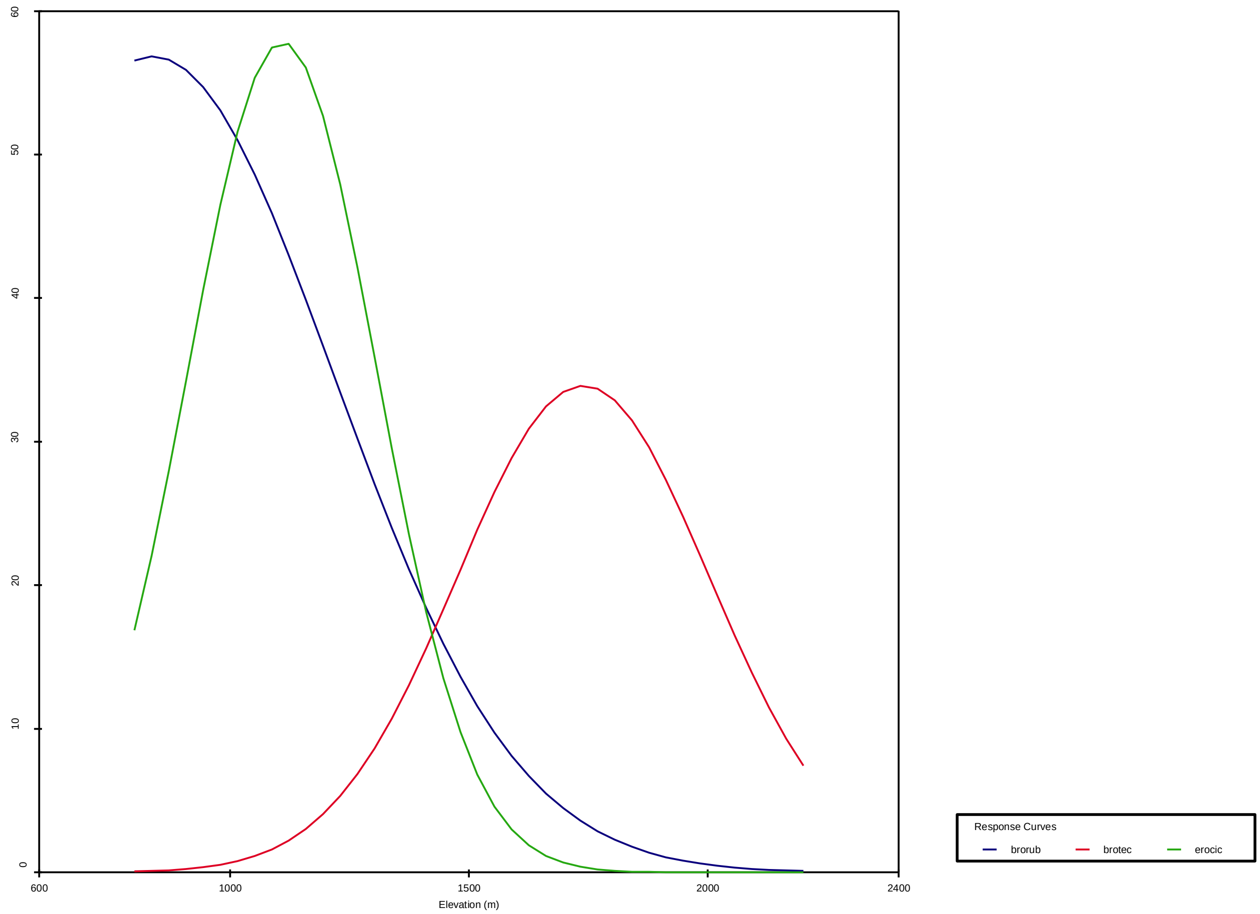


Figure 6-11. Response curves for *Bromus rubens* (brorub), *Bromus tectorum* (brotec) and *Erodium cicutarium* (erocic) along an elevation gradient in the Southern Nevada Complex fires of 2005.

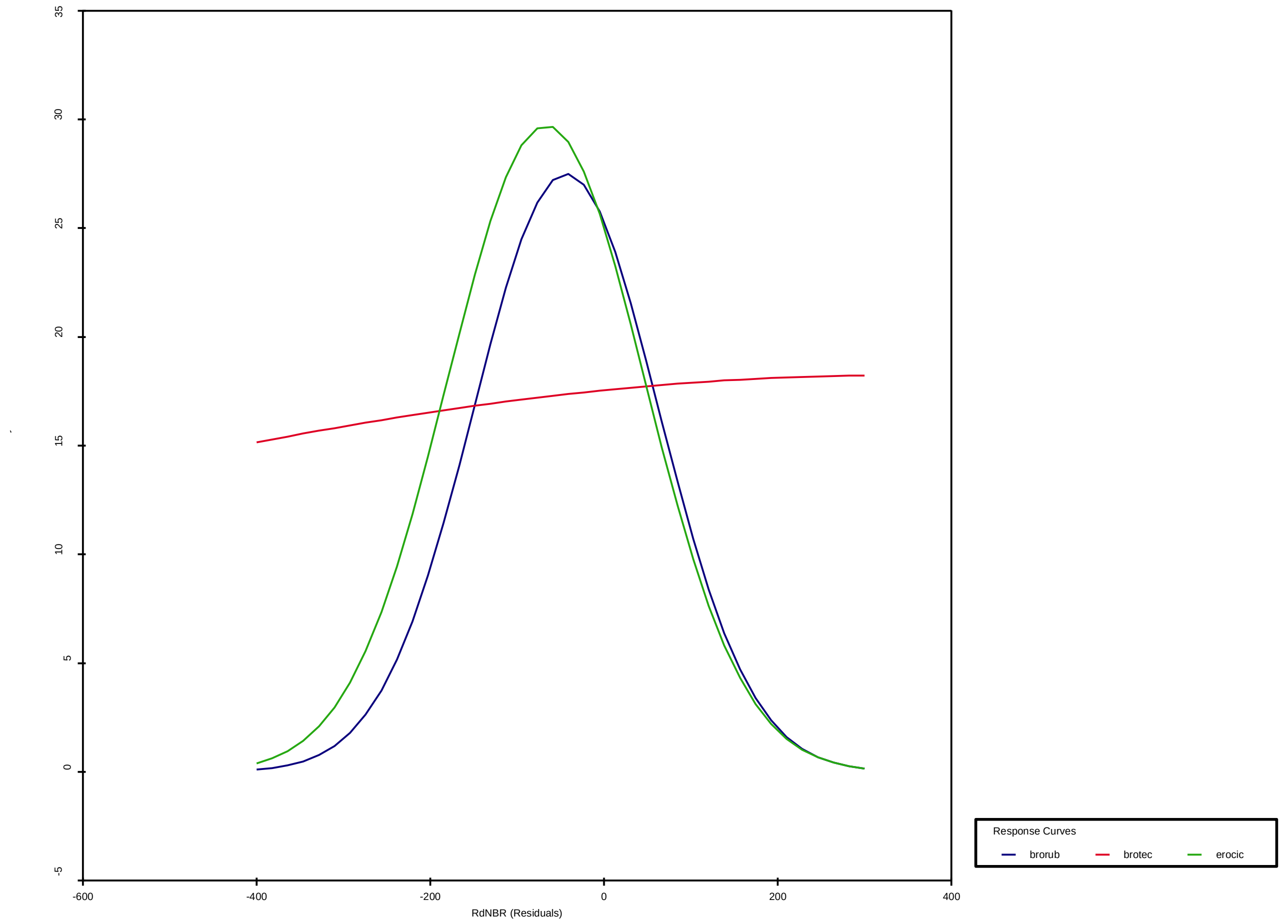


Figure 6-12. Response curves for *Bromus rubens* (brorub), *Bromus tectorum* (brotec) and *Erodium cicutarium* (erocic) along a fire severity gradient (RdNBR) in the Southern Nevada Complex fires of 2005. Residuals were used to remove potential confounding from a significant positive relationship between RdNBR and elevation.

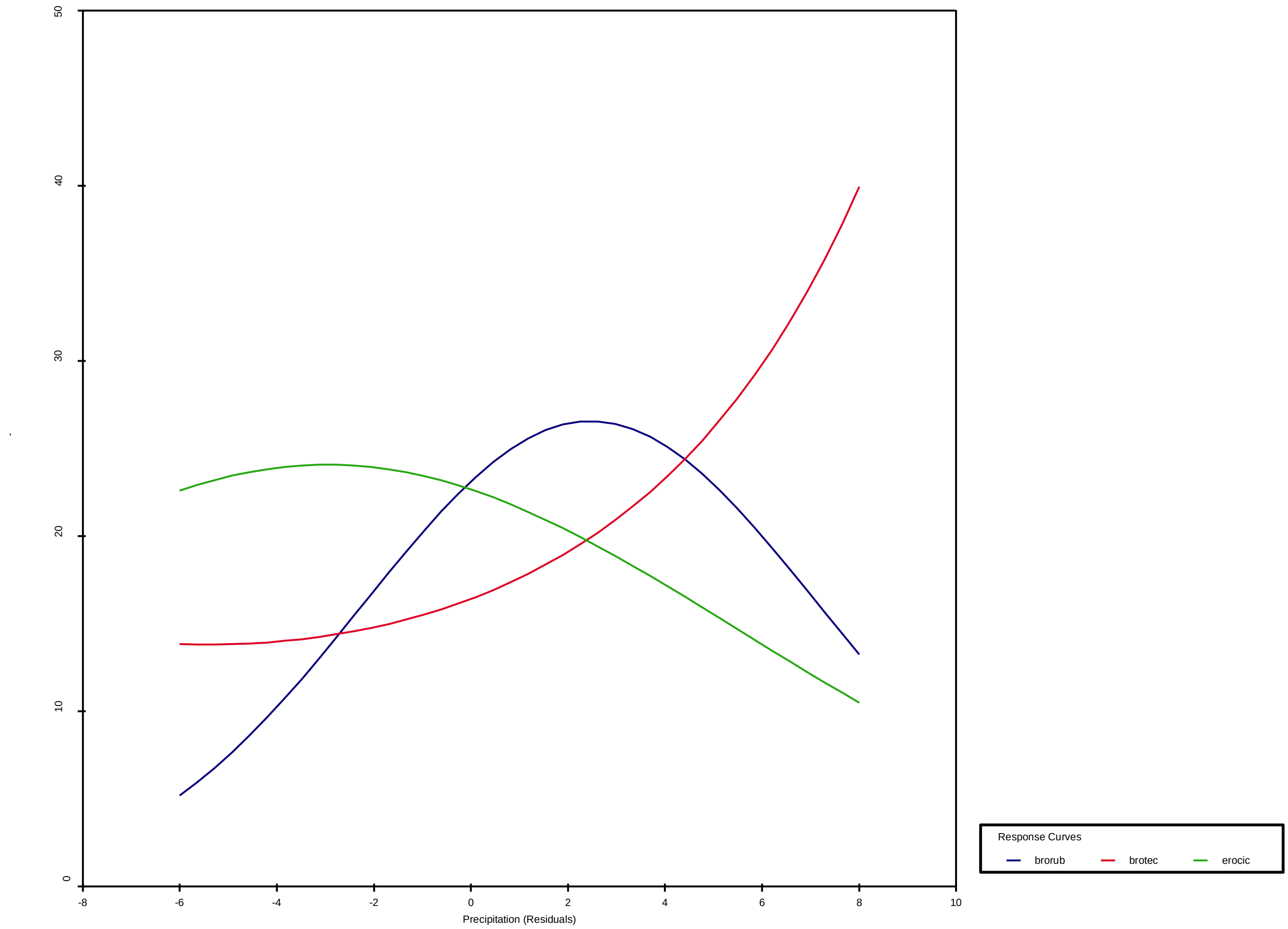


Figure 6-13. Response curves for *Bromus rubens* (brorub), *Bromus tectorum* (brotec) and *Erodium cicutarium* (erocic) along a precipitation gradient in the Southern Nevada Complex fires of 2005. Residuals were used to remove potential confounding from a significant positive relationship between precipitation and elevation.

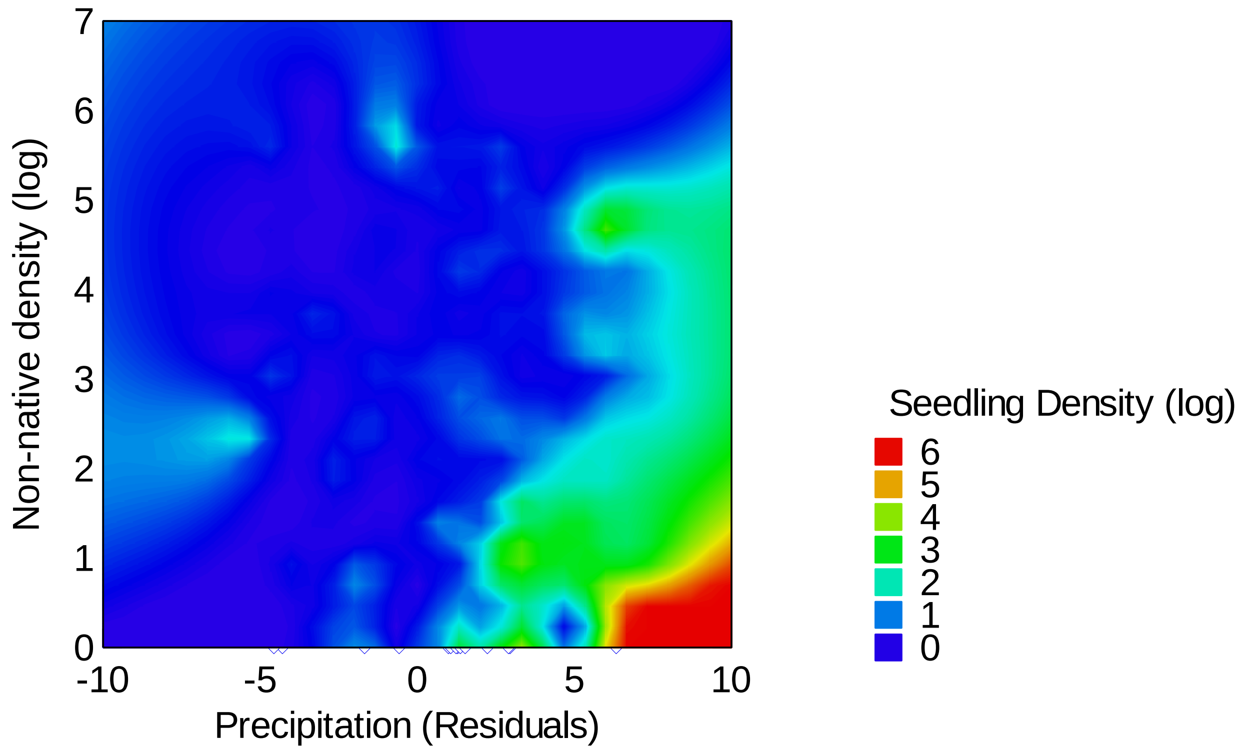


Figure 6-14. The relationship between density of woody seedlings, the pooled density of three non-native annual herbaceous species (*Bromus rubens*, *Bromus tectorum*, and *Erodium cicutarium*); non-native density, and precipitation in the Southern Nevada Complex fires of 2005. Residuals were used to remove potential confounding from a significant positive relationship between precipitation and elevation. Seedling and herbaceous species densities were $\log_e$ transformed.

Unburned Plots

Shrubs and trees comprised a mean of 32.1% (± 3.2 SE) of the vegetation cover in the unburned plots, while herbaceous species comprised a mean of 42.8% (± 3.9 SE). Six non-native species occurred in the unburned plots (*Bromus tectorum*, *B. rubens*, *Erodium cicutarium*, *Schismus barbatus*, *Sisymbrium altissimum*, and *S. irio*), and their density and cover were 1-3 orders of magnitude greater than that of the native herbaceous guilds (Table 6-8). *Bromus rubens* comprised 69% of the herbaceous density, while *B. tectorum* and *Erodium cicutarium* made up an additional 19% and 6%, respectively. Collectively, these three species also comprised 47.9% of relative cover (woody + herbaceous). These patterns were consistent among all vegetation types, although density of the non-native species was somewhat lower in natural regeneration vegetation than the other three types (Figure 6-15).

Table 6-8. The mean number of species (m^{-2}), density (individuals m^{-2}), and cover (%) for six herbaceous guilds in four unburned vegetation types adjacent to the Southern Nevada Complex fires of 2005. Vegetation in the "Natural Regeneration" category was Creosote/Thermic blackbrush. PJ = pinyon-juniper. Numbers in parentheses are ± SE.

Vegetation	Guild	Species	Density	Cover
Mesic Blackbrush	Native Annual Forb	0.90 (0.20)	8.7 (3.7)	0.7 (0.2)
	Native Perennial Forb	0.20 (0.10)	0.1 (0.1)	0.1
	Native Annual Grass	0.10 (0.10)	2.1 (1.6)	0.2 (0.1)
	Native Perennial Grass	0.10 (0.03)		
	Non-native Annual Grass	1.30 (0.10)	768.8 (188.5)	34.5 (3.7)
	Non-native Annual Forb	0.60 (0.10)	30.2 (8.5)	0.9 (0.3)
Natural Regeneration	Native Annual Forb	1.20 (0.20)	117.2 (63.6)	4.1 (1.6)
	Native Perennial Forb	0.50 (0.10)	4.8 (3.7)	0.6 (0.2)
	Native Annual Grass	0.10 (0.02)	3.9 (2.4)	0.4 (0.2)
	Native Perennial Grass	1.00 (0.10)	5.1 (2.5)	3.7 (1.0)
	Non-native Annual Grass	1.40 (0.10)	1172.0 (214.0)	40.1 (3.2)
	Non-native Annual Forb	0.60 (0.10)	72.5 (19.5)	1.8 (0.6)
Non-wilderness PJ	Native Annual Forb	0.70 (0.20)	2.2 (1.3)	1.4 (0.7)
	Native Perennial Forb	1.20 (0.20)	2.3 (0.8)	0.6 (0.2)
	Native Annual Grass	0.30 (0.20)	2.0 (1.7)	0.1 (0.1)
	Native Perennial Grass	0.80 (0.20)	2.0 (1.3)	1.7 (1.2)
	Non-native Annual Grass	1.70 (0.20)	357.5 (72.6)	32.0 (5.2)
	Non-native Annual Forb			0.1 (0.1)
Wilderness PJ	Native Annual Forb	0.40 (0.20)	0.1 (0.1)	
	Native Perennial Forb	1.80 (0.70)	2.9 (1.7)	0.8 (0.8)
	Native Annual Grass			
	Native Perennial Grass	0.80 (0.40)	0.7 (0.4)	1.7 (0.7)
	Non-native Annual Grass	2.20 (0.40)	805.2 (305.9)	38.0 (8.9)
	Non-native Annual Forb			
Overall Mean	Native Annual Forb	1.00 (0.10)	54.6 (28.1)	2.2 (0.7)
	Native Perennial Forb	0.50 (0.10)	2.6 (1.6)	0.4 (0.1)
	Native Annual Grass	0.10 (0.06)	2.8 (1.2)	0.3 (0.1)
	Native Perennial Grass	0.60 (0.10)	2.5 (1.1)	1.9 (0.5)
	Non-native Annual Grass	1.40 (0.10)	893.2 (121.4)	36.8 (2.1)
	Non-native Annual Forb	0.50 (0.10)	43.0 (9.5)	1.1 (0.3)

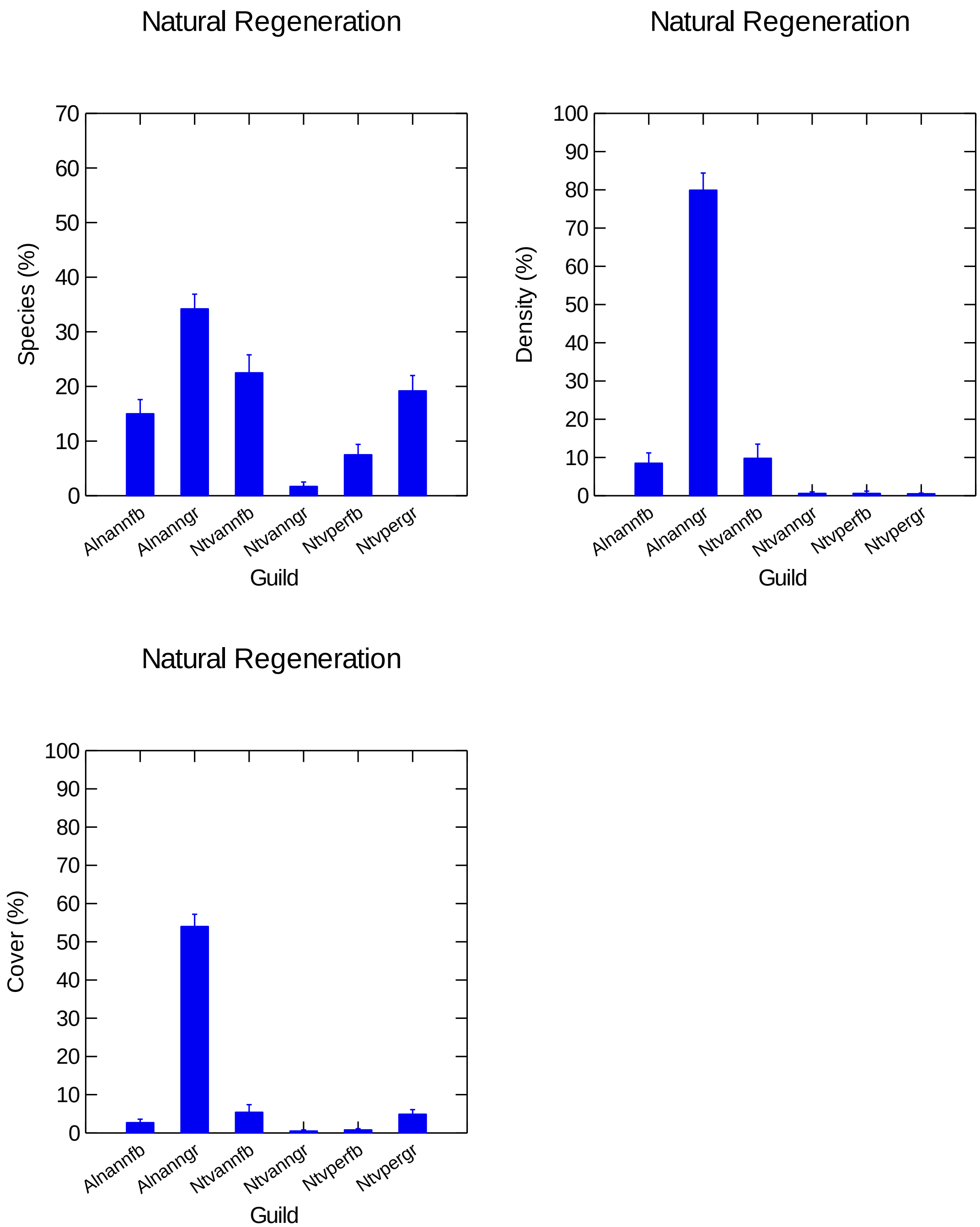


Figure 6-15. The percentage of species, relative density (percent individuals m^{-2}), and relative cover of six herbaceous vegetation guilds in four unburned vegetation types adjacent to the Southern Nevada Complex fires of 2005. The guilds were non-native annual forbs (Alnannfb), non-native annual grass (Alnanngr), native annual forbs (Ntvannfb), native annual grass (Ntvanngr), native perennial forbs (Ntvperfb), and native perennial grass (Ntvpergr). Vegetation in the "Natural Regeneration" category was Creosote/Thermic blackbrush. PJ = pinyon-juniper. The figure continues on the next three pages.

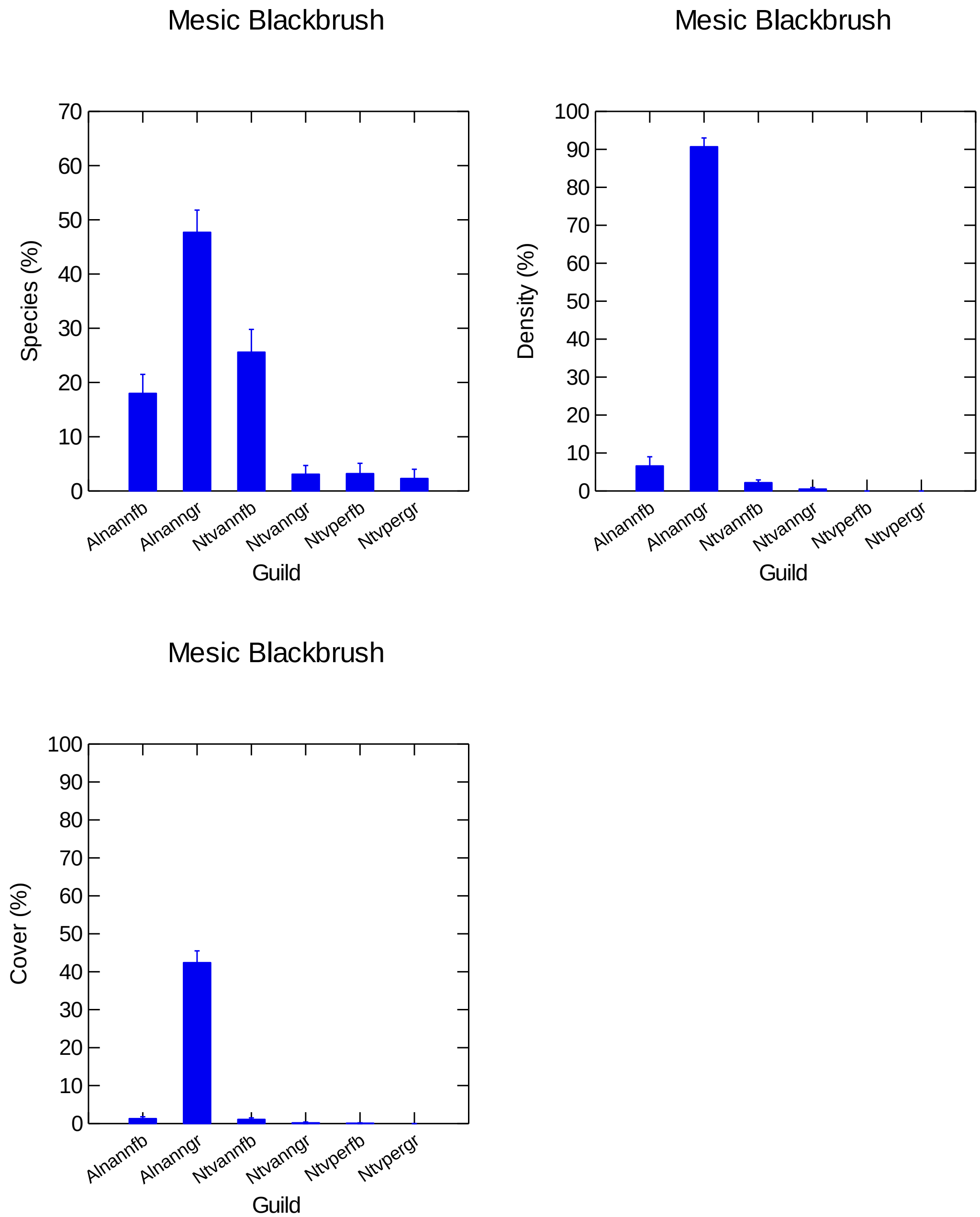
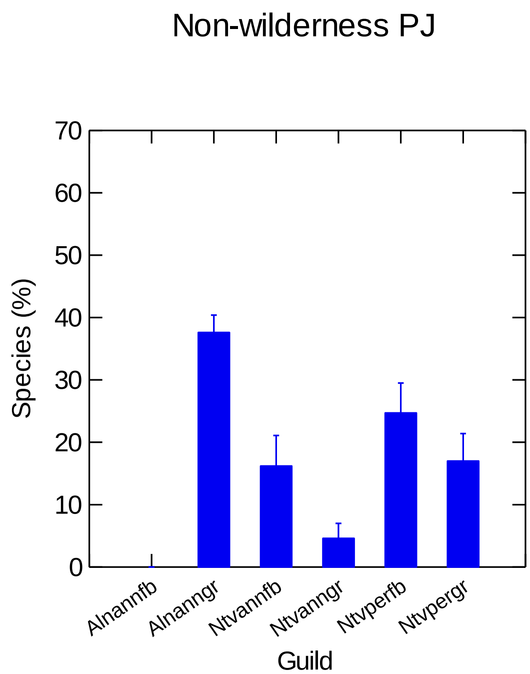
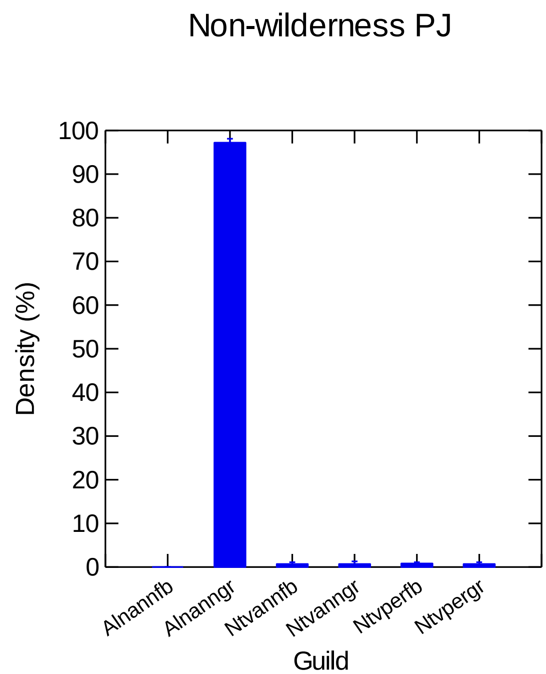
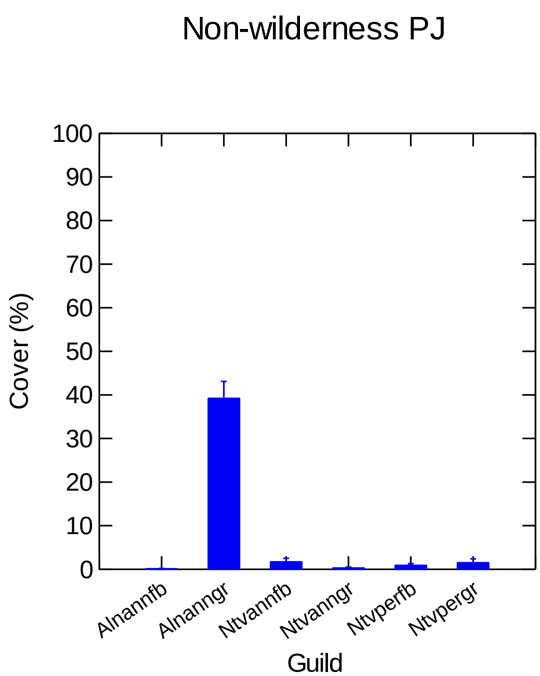


Figure 6-15 continued.







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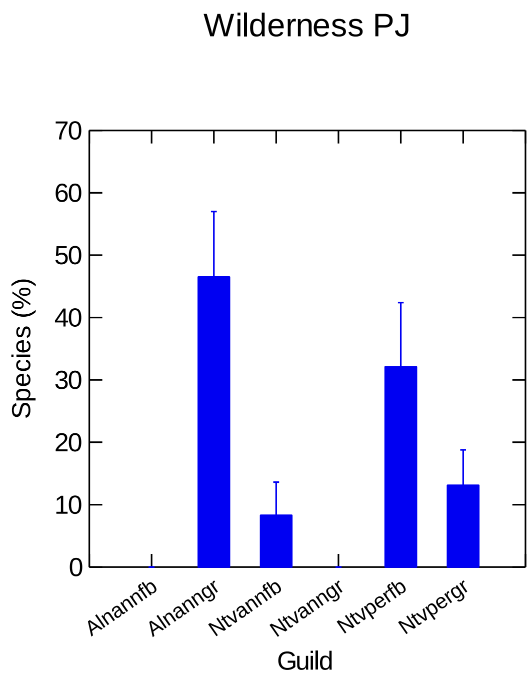

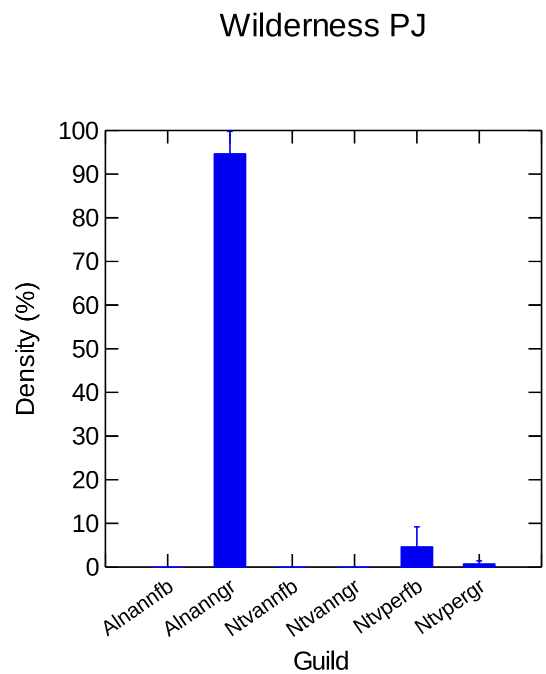

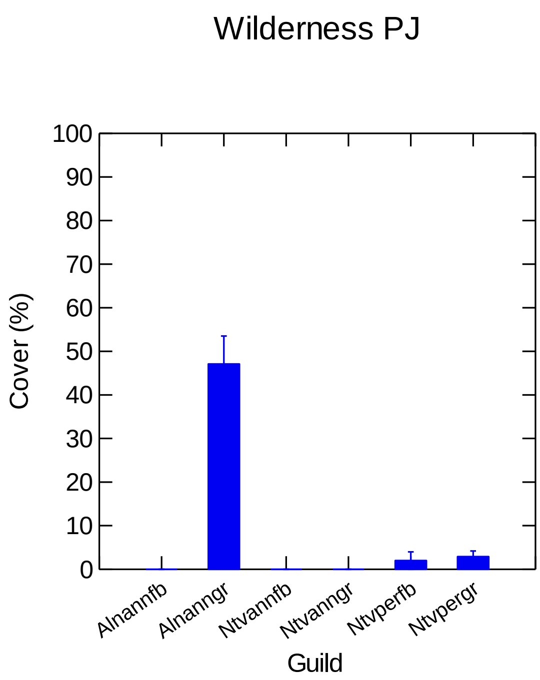

Figure 6-15 continued.

Cover

There was a strong relationship between cover and density for the five guilds of plants (Table 6-9). The correlation coefficients ranged from 0.761 to 0.924 and ΔAIC values from 1.25 to 28.44. Based on these values, we concluded that there was a reasonable justification for using density in the different vegetation types to estimate cover.

Non-native grasses and forbs dominated cover in natural regeneration and mesic blackbrush plots during all years, but were relatively less dominant in pinyon-juniper plots (Figure 6-16 and Table 6-10). The best supported model for all three herbaceous guilds indicated there were different patterns in variation over time among vegetation types (Appendix 6-7), but the best supported model for woody species indicated minimal differences in variation over time among vegetation types (Appendix 6-8). The absolute cover of non-native forbs was 3x-9x greater in natural regeneration and mesic blackbrush plots than pinyon-juniper plots (Figure 6-16). The absolute cover of non-native grasses in natural regeneration and mesic blackbrush plots decreased steadily over the three-year period of the study; however this decrease was compensated by an increase in cover of non-native forbs in the third year after the fires (Figure 6-16). Absolute cover of non-native grasses and forbs remained relatively similar across years in pinyon-juniper plots (Figure 6-16). The absolute cover of native herbaceous species was 2x-4x greater in pinyon-juniper plots than in natural regeneration and mesic blackbrush plots (Figure 6-16). There was little variation over time in cover of native herbaceous species except in mesic blackbrush plots (Appendix 6-8). There was a modest increase in cover of woody species in all vegetation types between the first and third years after the fires (Figure 6-16), however they never comprised more than 32% of the relative cover in any vegetation type in any year (Table 6-10).

Table 6-9. Summary statistics for the best supported model from a pool of eight candidate models for estimation of cover in five guilds of plants in the Southern Nevada Complex fires of 2005. Density is the mean number of individuals m^{-2} in each plot for the herbaceous guilds and the number of individuals 150 m^{-2} in the guild for shrubs and trees. There were four vegetation types, including "Natural Regeneration" (creosote/thermic blackbrush), mesic blackbrush, and wilderness and non-wilderness pinyon-juniper. ΔAIC is the difference in the Akaike Information Criterion between the best supported and next most supported models, r is the correlation coefficient between the observed and fitted values, and N the sample size (number of plots) used in the model.

Best Supported Model	**ΔAIC**	***r***	**N**
Non-native forb cover			
Constant + density + density^2 + vegetation + density*vegetation + density^2*vegetation	28.44	0.884	456
Non-native grass cover			
Constant + density + density^2 + vegetation	4.90	0.924	456
Native forb cover			
Constant + density + density^2 + vegetation + density*vegetation	1.25	0.859	456
Native grass cover			
Constant + density + density^2 + vegetation + density*vegetation + density^2*vegetation	6.79	0.899	456
Shrub and tree cover			
Constant + density + density^2 + vegetation + density*vegetation + density^2*vegetation	1.48	0.761	365

Table 6-10. Mean relative cover (% ± SE) of four groups of plants in four vegetation types that burned in the Southern Nevada Complex fires of 2005. Vegetation in the "Natural Regeneration" category was Creosote/Thermic blackbrush. PJ = pinyon-juniper. Sampling was not done in Natural Regeneration plots in the second year post-fire.

Years post-fire	Natural Regeneration	Mesic Blackbrush	Non-wilderness PJ	Wilderness PJ
Non-native Annual Forbs				
1	23.2 ± 3.3	39.2 ± 4.5	7.6 ± 3.4	4.6 ± 5.0
2		31.6 ± 9.4	7.0 ± 3.5	5.2 ± 5.2
3	53.0 ± 2.7	48.3 ± 5.0	8.6 ± 3.3	6.4 ± 4.4
Non-native Annual Grass				
1	62.0 ± 5.5	52.1 ± 6.9	51.0 ± 4.5	47.1 ± 7.7
2		46.2 ± 9.4	45.5 ± 4.7	41.6 ± 7.1
3	32.4 ± 4.0	33.0 ± 8.0	47.2 ± 4.7	47.0 ± 7.0
Native Forbs and Grasses				
1	5.5 ± 1.8	5.6 ± 2.1	18.1 ± 1.4	21.1 ± 2.3
2		12.3 ± 2.8	19.8 ± 1.5	21.9 ± 2.2
3	4.2 ± 1.3	10.3 ± 2.3	18.0 ± 1.4	18.5 ± 2.0
Shrubs and Trees				
1	9.3 ± 2.0	3.1 ± 2.8	23.2 ± 1.4	27.2 ± 2.7
2		9.9 ± 3.8	27.7 ± 1.5	31.2 ± 3.0
3	10.4 ± 1.7	8.4 ± 3.1	26.2 ± 1.4	28.2 ± 2.3

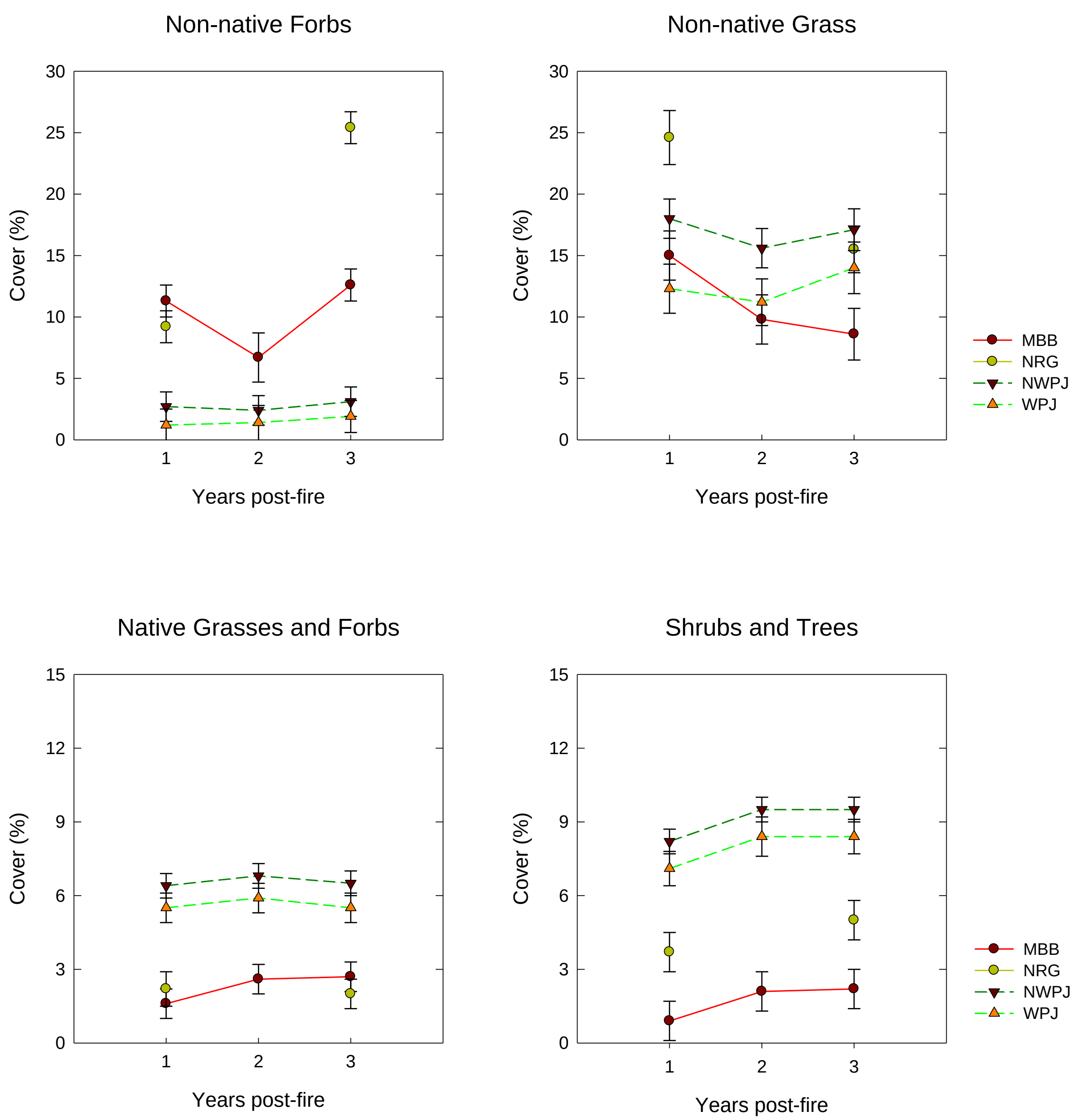

Figure 6-16. Absolute cover of herbaceous and woody species in four vegetation types burned in the Southern Nevada Complex fires of 2005. The vegetation types include mesic blackbrush scrub (MBB), natural regeneration (NRG; comprised of creosote and thermal blackbrush scrub), non-wilderness pinyon-juniper (NWPJ), and wilderness pinyon-juniper (WPJ). The first post-fire year was 2006. Lines are not shown for natural regeneration because sampling was not conducted in these plots in the second year post-burn. See text for methods used to derive cover estimates after the first year post-fire.

DISCUSSION

Post-fire succession patterns in the Southern Nevada Complex fires were shaped by a combination of abiotic and biotic factors. The abiotic factors included disturbance from fire and resource availability in the form of precipitation. Although we could not measure it directly, the overriding biotic factor appeared to be competition from three species of non-native annual herbs with seedlings of perennial plants.

It is likely that fire as a disturbance event, rather than variation in disturbance severity, was influencing vegetation patterns in the Southern Nevada Complex fires. This was particularly so in blackbrush and creosote communities, where rates of allogenic succession are known to be low (Brooks and Matchett 2003). We quantified disturbance amount using RdNBR, which is one of several widely used indices of fire severity (Miller and Thode 2007). An important advantage of RdNBR for measuring severity is that it corrects for differences in biomass between different vegetation types (more properly, differences in chlorophyll content among vegetation types; Miller et al. 2009) by quantifying relative changes rather than absolute changes due to fire. The CCA's indicated that RdNBR and elevation had a moderately strong positive correlation, however since the index corrects for bias due to initial vegetation biomass, it is likely that relative burn severity did not increase with elevation and was providing unique information on species distribution and abundance patterns. It is also likely that the two variables represent different influences on vegetation patterns. RdNBR measures disturbance amount while elevation likely integrates several factors influencing plant growth, such as light availability, temperature, and soil moisture availability. The influence of fire severity though, at least as measured by RdNBR, appeared to be more important for individual species rather than entire guilds. For example, density of the shrubs *Coleogyne ramosissima*, *Encelia virginensis*, and *Thamnosma montana* was much higher in sites with low values of RdNBR, whereas density of *Ericameria nauseosa* and *Ephedra viridis* was high in sites where values of RdNBR were high. However, there was no relationship between RdNBR and the herbaceous or woody guilds.

Precipitation had an extremely important influence on succession patterns and species composition in the vegetation communities studied. Precipitation has long been recognized to be the primary limiting resource for plants in arid environments (e.g. Brooks 1999, 2000, DeFalco et al. 2007), and while availability of other resources (e.g. nutrients; Brooks 2003) may have also been a factor influencing succession patterns, the importance of precipitation was clear. However, it was not just the amount of rainfall that occurred in a given year, but the timing of it as well. Species composition in all of the vegetation types was affected dramatically by the amount of rainfall that occurred in the early or latter periods of the wet season.

Competition can be one of the most important influences on the local and neighborhood structure of plant communities, especially in the Mojave Desert (Brooks 2003, DeFalco et al. 2003, 2007). Although they comprised less than 10% of the herbaceous flora, stems of non-native grasses and forbs dominated post-fire vegetation communities in the Southern Nevada Complex fires. The overwhelming contribution to these patterns was by three non-native annual species: cheatgrass, red brome, and *Erodium*. Collectively, these three species made up 90% or more of the stems in the plots. There was evidence that competition from these three species was intense enough that it was likely suppressing growth of native woody seedlings and species richness of native perennial grasses. However, species richness of annuals, especially natives, increased during the study, and stem density and species richness of native forbs co-varied positively with stem density of the two Bromes and *Erodium*. This likely reflected a common response to an increase in resource availability, regardless of whether species were native or not (Stohlgren et al. 1999). Other studies that have reported negative relationships between abundances of native and non-native annuals (e.g. Brooks 2003, DeFalco et al. 2003, 2007) were focused on late successional plant communities where resource competition may have been more acute. In addition, it is important to note that stem density of other species was very low compared to that of the two Bromes and *Erodium*. Moreover, the gradient in stem density of cheatgrass, red brome, and *Erodium* was not particularly steep; in essence, whether individually or cumulatively, the three species were uniformly abundant throughout the vegetation communities. Therefore, it is difficult to say whether richness and stem densities of other species would increase if stem density of the two Bromes and *Erodium* was substantially reduced. If however there was a true positive association between native and non-native annuals, then the two Bromes and *Erodium* could be appropriately considered “drivers” of change in the vegetation types (MacDougall and Turkington 2005). If density and/or species richness of other species did not increase though, these three species are likely “passengers” that are responding to a disturbed environment (MacDougall and Turkington 2005). The competitive effect of cheatgrass, red brome, and *Erodium* was expressed primarily by suppressing abundance of other species, which is why three of the four diversity indices (N_1, N_2, and $E_{1/d}$) and woody seedling density had strong negative relationships with stem density of the two Bromes and *Erodium*.

Despite their dominance of the vegetation, the patterns in distribution and abundance of cheatgrass, red brome, and *Erodium* were highly dynamic. The two grasses dominated the vegetation communities during the first year after the fires, but then dramatically decreased during the next two years. However, the decrease in the grasses was compensated by a very large increase in density of *Erodium*. Moreover, due to their broad overlapping distributions, the species not only exerted individual effects but a strong cumulative effect as well. As a consequence, the post-fire flora was dominated by one or more of these species regardless of year, elevation zone, vegetation type, or burned area.

Fire severity as indexed by RdNBR had virtually no influence on abundance patterns of the Brome grasses or *Erodium*. Caution needs to be applied when interpreting this pattern though. Without pre-burn data or unburned control plots it is difficult to determine what the degree of change in abundance of the Brome grasses or *Erodium* was following the fires. Most of the plots occurred in sites that had not previously burned in many decades, implying that the Bromes and *Erodium* were at least in the seed bank, and may very well have comprised a significant proportion of the aboveground vegetation. RdNBR indexes relative change, and without a similar measure of change from plot-based sampling it would be premature to assume there is no relationship between fire severity based on RdNBR and dominance of non-native annuals.

While precipitation had a strong positive effect on both native and non-native annual plant species, the relative importance of precipitation was not uniformly consistent for cheatgrass, red brome, and *Erodium*. Red brome and *Erodium* had higher abundances in areas with low or, at most, average amounts of precipitation. In contrast, cheatgrass abundance increased dramatically as precipitation increased. This presents a profound problem for management, because different non-native species can proliferate under different rainfall regimes (years). Post-fire seeding in dry years may not have any greater chance of seedling germination and establishment than seeding in wet years (or vice-versa) because non-natives may dominate in all years, regardless of what the specific identity of any dominant non-native in any particular year is. It may behoove management to not just do blanket seeding, but target specific vegetation types (or, what is likely a more ecologically relevant approach, specific elevation zones) in years with a particular rainfall pattern (e.g. seed higher elevation areas in dry years but not wet years), and/or limit seeding to when field examinations show low populations of annual grasses or forbs.

Precipitation was not an important factor in regeneration of woody stems, but it was an extremely important factor in woody seedling germination. However, it was additionally dependent on the level of dominance by Brome grass and *Erodium*; once density of these species exceeded 100-150 stems per m^2 there was little if any germination of woody seeds. This is extremely important information for management; not only is post-fire seeding of herbaceous species going to be, in all likelihood, unsuccessful in areas dominated by non-native annual species, but seeding of woody species would be no more likely to succeed than seeding herbaceous species. Moreover, it is possible that at this point vegetation communities will be set on a trajectory to an alternative state where the community is characterized by non-native annual herbaceous species rather than native woody and perennial herbaceous species.

The results from the first three postfire years in the Southern Nevada Complex fires of 2005 suggest that post-burn vegetation communities in the Mojave Desert could undergo transitions to alternative states characterized by high levels of non-native annual plant density and cover and low levels of perennial plant density and cover even after a single fire event. Many areas in the Southern Nevada Complex fires of 2005 were likely pre-disposed to such a conversion as non-native species, especially *Bromus rubens* and *B. tectorum*, dominated the herbaceous layer in the unburned plots. Following the fires, woody regeneration by seedling and resprouts appeared to be strongly limited by non-native annual grasses and forbs. This suppression of woody regeneration could be enough to allow the communities to remain dominated by non-native annual species over time. Alternatively, it may be if these areas do not undergo further disturbances for several decades, or regeneration by woody species is not impeded by grazing, then succession could lead to shrub-dominated communities. The short duration of the study and lack of unburned controls make this impossible to test with the current data set, but it provides a hypothesis-generating framework for future study.

The types of analyses that we could conduct with the cover data were limited because we had to estimate cover by guilds rather than species. Nevertheless, interpretations of succession patterns were largely consistent between analyses conducted with either density or cover data; cover tended to be dominated by non-native grasses and forbs in all vegetation types, especially at lower elevations. However, cover of non-native herbs varied spatially and temporally. Non-native forb cover was greater in lower elevation communities, while non-native grass cover tended to be greater in higher elevation communities. Cover of non-native grasses and forbs varied more in mesic blackbrush than other vegetation communities, but it was unclear why this was so. The significant aspect of the temporal variation of non-native guilds in mesic blackbrush was that decreased cover of grasses was offset by increased forb cover.

The greatest increase in shrub and tree cover occurred primarily in the second year after the fires. Although woody cover appeared to reach an asymptote in the third year after the fire, it would be premature to assume that it will not continue to increase for several more years, albeit at relatively slow rates. Woody regeneration following fire in arid communities tends to be slower than that following fire in more mesic communities, but these slow rates may be exacerbated by the suppression of woody seedling regeneration by non-native annual species. It is likely that woody cover in the burned areas will be comprised of resprouting individuals, and that a significant proportion of the cover in the understory will be comprised of non-native annual species.

In terms of future work, there should be greater emphasis placed not just on describing patterns following fires and post-fire management activities, but on integration of approaches that allow a better mechanistic understanding of what produced the patterns. This

would be of tremendous benefit to management; if there is an understanding of *why* and *how* something occurred, not just *what* occurred, then managers will be able to better target when and where future actions will have the greatest probability of success.

An important first step towards gaining a better understanding of post-fire succession patterns in the Mojave Desert would be including unburned plots into the study design. Preburn plots would be ideal, but the unpredictable nature of fire makes it unrealistic to expect to know where to locate such plots. However, unburned plots in similar vegetation near burned areas are a legitimate and important alternative, and would be extremely helpful for making comparisons between burned and unburned conditions. Though they were only sampled in 2006, the unburned plots in the current study provided valuable insight on what vegetation structure and species composition may have been liked in the burned areas if the 2005 fires had not occurred. If these plots had been sampled throughout the duration of the study, then the succession trajectories we analyzed in the burned areas could have been compared to those in unburned areas.

A second step would be collecting seed bank data in future projects. As mentioned above, all indications are that the Brome grasses and *Erodium* were significant components of the aboveground vegetation, the seed bank, or both. Unburned reference sites and seed bank samples would have been a relatively easy and inexpensive way of quantifying this.

Another important way to integrate a better understanding of pattern and process into future projects is inclusion of small scale experimental studies focused on identifying the forces that are likely driving large scale patterns of change. Our analysis indicates that competition from the Brome grasses and *Erodium* may have been the principal mechanism responsible for suppression of regeneration by woody species and native bunchgrasses in the burns, while precipitation was the primary factor responsible for higher levels of recruitment. However, while precipitation was related to increased species richness and stem density of native and non-native annual species, competition from the Brome grasses and *Erodium* was not a factor in suppressing regeneration of native herbaceous species. Small scale field and/or greenhouse competition and resource availability experiments would have allowed us to better evaluate not just what the post-fire succession patterns were, but the relative importance of different mechanisms producing the patterns.

MANAGEMENT IMPLICATIONS

Burn severity as measured by RdNBR was not consistently associated with vegetation responses. The density of some species was highest at low RdNBR values, whereas others were highest at high RdNBR values.

Annual and seasonal precipitation had an overriding effect on the species composition of plant communities during the three postfire years. In addition, establishment of seedlings of woody species only occurred where rainfall as relatively high.

It appeared that that non-native annuals red brome, cheatgrass, and *Erodium* may be been competing with and suppressing the establishment and/or growth of native perennial plants. Although the dominance of these three species varied across the elevational gradient and among years of contrasting rainfall, there was always at least one of the species dominating the landscape in most every situation. Only on the rare occasion that density of these species was low and rainfall as high, were seedlings of woody perennials were abundant.

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Appendix 6-1. Model selection statistics for five herbaceous vegetation response variables in four areas (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to "natural regeneration" sites (creosote and thermic blackbrush vegetation). ΔAICc is the difference in the bias-corrected Akaike Information Criterion (AICc) between a given model and the best supported model, wAICc is the absolute support for a given model (= exp(-ΔAICc/2)), and rwAICc is the support relative to the other models.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Total stem density (log)					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	25.36	0.0000	0.0000
	5	Model 4 + vegetation type	60.44	0.0000	0.0000
	4	Model 3 + year^2	285.19	0.0000	0.0000
	2	Random intercept - fire	304.04	0.0000	0.0000
	3	Model 2 + year	304.20	0.0000	0.0000
	1	Null - fixed intercept	307.19	0.0000	0.0000
N_0					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	37.85	0.0000	0.0000
	5	Model 4 + vegetation type	68.56	0.0000	0.0000
	4	Model 3 + year^2	105.69	0.0000	0.0000
	3	Model 2 + year	252.83	0.0000	0.0000
	1	Null - fixed intercept	393.21	0.0000	0.0000
	2	Random intercept - fire	394.95	0.0000	0.0000
N_1					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	0.9400
	6	Model 5 + year*vegetation type	5.50	0.0639	0.0600
	5	Model 4 + vegetation type	37.84	0.0000	0.0000
	4	Model 3 + year^2	64.80	0.0000	0.0000
	3	Model 2 + year	102.85	0.0000	0.0000
	2	Random intercept - fire	142.31	0.0000	0.0000
	1	Null - fixed intercept	143.97	0.0000	0.0000
N_2					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	0.6972
	6	Model 5 + year*vegetation type	1.67	0.4342	0.3028
	5	Model 4 + vegetation type	44.91	0.0000	0.0000
	4	Model 3 + year^2	57.53	0.0000	0.0000
	3	Model 2 + year	80.54	0.0000	0.0000
	2	Random intercept - fire	98.76	0.0000	0.0000
	1	Null - fixed intercept	99.33	0.0000	0.0000

Appendix 6-1 continued.

Model		**Variables**	**ΔAICc**	**exp(ΔAICc)**	**wAICc**
Evenness[$E_{1/d}$/(S)]					
	6	Model 5 + year*vegetation type	0.00	1.0000	0.5944
	7	Model 6 + year^2*vegetation type	0.76	0.6823	0.4056
	1	Null - fixed intercept	38.05	0.0000	0.0000
	2	Random intercept - fire	39.80	0.0000	0.0000
	3	Model 2 + year	41.76	0.0000	0.0000
	4	Model 3 + year^2	43.07	0.0000	0.0000
	5	Model 4 + vegetation type	45.96	0.0000	0.0000

Appendix 6-2. Parameters from the best supported models for four indices of herbaceous species diversity in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to "natural regeneration" sites (creosote and thermic blackbrush vegetation). Percent is the variation (%) attributable to the random factor of Fire (Delamar, Duzak, Halfway, and Meadow Valley).

Source of variation	Parameter	S.E.	Z	P	Percent
N0					
Fixed effects					
Constant	6.378	0.412	15.481	0.000	
Year	-6.606	0.945	6.990	0.000	
Year^2	4.458	0.436	10.225	0.000	
MBB	-1.325	0.504	2.629	0.004	
NWPJ	0.177	0.465	0.381	0.352	
WPJ	-0.113	0.546	0.207	0.418	
Year*MBB	1.501	1.236	1.214	0.112	
Year*NWPJ	4.523	1.100	4.112	0.000	
Year*WPJ	5.017	1.323	3.792	0.000	
Year^2*MBB	-0.940	0.585	1.607	0.045	
Year^2*NWPJ	-3.034	0.515	5.891	0.000	
Year^2*WPJ	-2.710	0.624	4.343	0.000	
Random effect					
Fire	0.001	0.001	1.000	0.159	0.0
N1					
Fixed effects					
Constant	2.135	0.190	11.237	0.000	
Year	-1.197	0.403	2.970	0.001	
Year^2	0.904	0.186	4.860	0.000	
MBB	0.268	0.221	1.213	0.113	
NWPJ	0.087	0.199	0.437	0.331	
WPJ	0.337	0.234	1.440	0.075	
Year*MBB	-0.501	0.526	0.952	0.171	
Year*NWPJ	1.101	0.469	2.348	0.009	
Year*WPJ	0.724	0.563	1.286	0.099	
Year^2*MBB	-0.138	0.249	0.554	0.290	
Year^2*NWPJ	-0.645	0.219	2.945	0.002	
Year^2*WPJ	-0.491	0.266	1.846	0.032	
Random effect					
Fire	0.021	0.020	1.050	0.147	1.8

Appendix 6-2 continued.

Source of variation	**Parameter**	**S.E.**	**Z**	**P**	**Percent**
N2					
Fixed effects					
Constant	1.715	0.144	11.910	0.000	
Year	-0.644	0.310	2.077	0.019	
Year^2	0.534	0.143	3.734	0.000	
MBB	0.362	0.170	2.129	0.017	
NWPJ	0.089	0.153	0.582	0.280	
WPJ	0.288	0.180	1.600	0.055	
Year*MBB	-0.564	0.405	1.393	0.082	
Year*NWPJ	0.649	0.361	1.798	0.036	
Year*WPJ	0.274	0.433	0.633	0.263	
Year^2*MBB	-0.053	0.192	0.276	0.391	
Year^2*NWPJ	-0.392	0.169	2.320	0.010	
Year^2*WPJ	-0.283	0.204	1.387	0.083	
Random effect					
Fire	0.009	0.010	0.900	0.184	1.3
H/log(N)					
Fixed effects					
Constant	0.409	0.032	12.781	0.000	
Year	0.013	0.039	0.333	0.370	
Year^2	0.011	0.015	0.733	0.232	
MBB	0.117	0.039	3.000	0.001	
NWPJ	-0.002	0.035	0.057	0.477	
WPJ	0.064	0.042	1.524	0.064	
Year*MBB	-0.149	0.029	5.138	0.000	
Year*NWPJ	0.011	0.026	0.423	0.336	
Year*WPJ	-0.048	0.031	1.548	0.061	
Random effect					
Fire	0.001	0.001	1.000	0.159	2.2

Appendix 6-3. Model selection statistics for four response variables for each of six herbaceous vegetation guilds in four areas (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to “natural regeneration” sites (creosote and thermic blackbrush vegetation). ΔAICc is the difference in the bias-corrected Akaike Information Criterion (AICc) between a given model and the best supported model, wAICc is the absolute support for a given model (= exp(-ΔAICc/2)), and rwAICc is the support relative to the other models.

Model		Variables	ΔAICc	wAICc	rwAICc
Non-native Annual Grass Stem density (log)					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	28.49	0.0000	0.0000
	5	Model 4 + vegetation type	42.03	0.0000	0.0000
	4	Model 3 + year^2	55.38	0.0000	0.0000
	3	Model 2 + year	91.49	0.0000	0.0000
	2	Random intercept - fire	129.02	0.0000	0.0000
	1	Null - fixed intercept	135.88	0.0000	0.0000
Non-native Annual Grass Stem density (%)					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	22.53	0.0000	0.0000
	5	Model 4 + vegetation type	78.02	0.0000	0.0000
	4	Model 3 + year^2	258.23	0.0000	0.0000
	3	Model 2 + year	263.32	0.0000	0.0000
	2	Random intercept - fire	367.90	0.0000	0.0000
	1	Null - fixed intercept	389.45	0.0000	0.0000
Non-native Annual Grass Species (%)					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	0.9956
	6	Model 5 + year*vegetation type	10.83	0.0045	0.0044
	5	Model 4 + vegetation type	68.81	0.0000	0.0000
	4	Model 3 + year^2	85.85	0.0000	0.0000
	3	Model 2 + year	114.74	0.0000	0.0000
	2	Random intercept - fire	244.04	0.0000	0.0000
	1	Null - fixed intercept	275.94	0.0000	0.0000

Appendix 6-3 continued.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Non-native Annual Forbs Stem density (log)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	24.28	0.0000	0.0000
	5	Model 4 + vegetation type	113.61	0.0000	0.0000
	3	Model 2 + year	859.20	0.0000	0.0000
	4	Model 3 + $year^2$	861.18	0.0000	0.0000
	2	Random intercept - fire	939.67	0.0000	0.0000
	1	Null - fixed intercept	1002.21	0.0000	0.0000
Non-native Annual Forbs Stem density (%)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	97.15	0.0000	0.0000
	5	Model 4 + vegetation type	170.21	0.0000	0.0000
	4	Model 3 + $year^2$	716.94	0.0000	0.0000
	3	Model 2 + year	726.46	0.0000	0.0000
	2	Random intercept - fire	835.24	0.0000	0.0000
	1	Null - fixed intercept	917.84	0.0000	0.0000
Non-native Annual Forbs Species (%)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	68.04	0.0000	0.0000
	5	Model 4 + vegetation type	103.27	0.0000	0.0000
	4	Model 3 + $year^2$	387.05	0.0000	0.0000
	3	Model 2 + year	398.72	0.0000	0.0000
	2	Random intercept - fire	402.64	0.0000	0.0000
	1	Null - fixed intercept	409.44	0.0000	0.0000
Native Annual Forbs Stem density (log)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	40.51	0.0000	0.0000
	5	Model 4 + vegetation type	95.24	0.0000	0.0000
	3	Model 3 + $year^2$	168.01	0.0000	0.0000
	4	Model 2 + year	290.32	0.0000	0.0000
	2	Null - fixed intercept	431.25	0.0000	0.0000
	1	Random intercept - fire	432.48	0.0000	0.0000

Appendix 6-3 continued.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Native Annual Forbs					
Stem density (%)					
	6	Model 5 + year*vegetation type	0.00	1.0000	0.5200
	7	Model 6 + $year^2$*vegetation type	0.38	0.8251	0.4290
	5	Model 4 + vegetation type	4.64	0.0982	0.0511
	4	Model 3 + $year^2$	48.94	0.0000	0.0000
	3	Model 2 + year	116.35	0.0000	0.0000
	2	Random intercept - fire	214.69	0.0000	0.0000
	1	Null - fixed intercept	215.50	0.0000	0.0000
Native Annual Forbs					
Species					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	46.67	0.0000	0.0000
	5	Model 4 + vegetation type	86.81	0.0000	0.0000
	4	Model 3 + $year^2$	241.53	0.0000	0.0000
	1	Null - fixed intercept	444.12	0.0000	0.0000
	2	Random intercept - fire	446.14	0.0000	0.0000
	3	Model 2 + year	448.15	0.0000	0.0000
Native Annual Forbs					
Species (%)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	23.41	0.0000	0.0000
	5	Model 4 + vegetation type	40.62	0.0000	0.0000
	4	Model 3 + $year^2$	58.82	0.0000	0.0000
	3	Model 2 + year	190.56	0.0000	0.0000
	1	Null - fixed intercept	270.42	0.0000	0.0000
	2	Random intercept - fire	272.44	0.0000	0.0000
Native Perennial Forbs					
Stem density (log)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	82.63	0.0000	0.0000
	5	Model 4 + vegetation type	126.53	0.0000	0.0000
	3	Model 2 + year	167.09	0.0000	0.0000
	4	Model 3 + $year^2$	168.68	0.0000	0.0000
	2	Random intercept - fire	215.09	0.0000	0.0000
	1	Null - fixed intercept	220.33	0.0000	0.0000

Appendix 6-3 continued.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Native Perennial Forbs Stem density (%)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	29.41	0.0000	0.0000
	5	Model 4 + vegetation type	32.87	0.0000	0.0000
	3	Model 2 + year	171.44	0.0000	0.0000
	4	Model 3 + $year^2$	182.74	0.0000	0.0000
	2	Random intercept - fire	194.66	0.0000	0.0000
	1	Null - fixed intercept	211.11	0.0000	0.0000
Native Perennial Forbs Species					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	81.02	0.0000	0.0000
	5	Model 4 + vegetation type	161.84	0.0000	0.0000
	4	Model 3 + $year^2$	173.97	0.0000	0.0000
	3	Model 2 + year	234.38	0.0000	0.0000
	2	Random intercept - fire	242.56	0.0000	0.0000
	1	Null - fixed intercept	297.39	0.0000	0.0000
Native Perennial Forbs Species (%)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	1.0000
	6	Model 5 + year*vegetation type	21.31	0.0000	0.0000
	5	Model 4 + vegetation type	117.92	0.0000	0.0000
	3	Model 2 + year	299.36	0.0000	0.0000
	4	Model 3 + $year^2$	300.45	0.0000	0.0000
	2	Random intercept - fire	310.29	0.0000	0.0000
	1	Null - fixed intercept	340.29	0.0000	0.0000
Native Annual Grass Stem density (log)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	0.4929
	5	Model 4 + vegetation type	0.49	0.7819	0.3854
	6	Model 5 + year*vegetation type	2.89	0.2359	0.1163
	4	Model 3 + $year^2$	9.01	0.0110	0.0054
	3	Model 2 + year	34.30	0.0000	0.0000
	1	Null - fixed intercept	104.51	0.0000	0.0000
	2	Random intercept - fire	106.52	0.0000	0.0000

Appendix 6-3 continued.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Native Annual Grass Stem density (%)					
	4	Model 3 + year^2	0.00	1.0000	0.8502
	5	Model 4 + vegetation type	3.91	0.1416	0.1204
	6	Model 5 + year*vegetation type	7.48	0.0237	0.0202
	7	Model 6 + year^2*vegetation type	9.04	0.0109	0.0093
	3	Model 2 + year	20.31	0.0000	0.0000
	1	Null - fixed intercept	99.07	0.0000	0.0000
	2	Random intercept - fire	101.08	0.0000	0.0000
Native Annual Grass Species					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	0.9205
	5	Model 4 + vegetation type	5.78	0.0555	0.0511
	6	Model 5 + year*vegetation type	6.96	0.0308	0.0283
	4	Model 3 + year^2	18.94	0.0001	0.0001
	3	Model 2 + year	61.13	0.0000	0.0000
	1	Null - fixed intercept	119.44	0.0000	0.0000
	2	Random intercept - fire	121.45	0.0000	0.0000
Native Annual Grass Species (%)					
	7	Model 6 + year^2*vegetation type	0.00	1.0000	0.6694
	5	Model 4 + vegetation type	1.59	0.4516	0.3023
	6	Model 5 + year*vegetation type	6.33	0.0423	0.0283
	4	Model 3 + year^2	19.61	0.0001	0.0000
	3	Model 2 + year	60.75	0.0000	0.0000
	1	Null - fixed intercept	146.11	0.0000	0.0000
	2	Random intercept - fire	148.13	0.0000	0.0000
Native Perennial Grass Stem density (log)					
	5	Model 4 + vegetation type	0.00	1.0000	0.5721
	6	Model 5 + year*vegetation type	1.59	0.4513	0.2582
	2	Random intercept - fire	4.34	0.1139	0.0652
	3	Model 2 + year	4.75	0.0928	0.0531
	7	Model 6 + year^2*vegetation type	5.97	0.0504	0.0288
	4	Model 3 + year^2	6.76	0.0340	0.0195
	1	Null - fixed intercept	10.36	0.0056	0.0032

Appendix 6-3 continued.

Model	**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Native Perennial Grass Stem density (%)				
5	Model 4 + vegetation type	2.29	0.3185	0.2043
6	Model 5 + year*vegetation type	3.03	0.2195	0.1408
7	Model 6 + $year^2$*vegetation type	7.77	0.0206	0.0132
2	Random intercept - fire	20.15	0.0000	0.0000
3	Model 2 + year	20.91	0.0000	0.0000
4	Model 3 + $year^2$	22.06	0.0000	0.0000
1	Null - fixed intercept	38.57	0.0000	0.0000
Native Perennial Grass Species				
5	Model 4 + vegetation type	0.00	1.0000	0.7304
6	Model 5 + year*vegetation type	2.25	0.3246	0.2371
7	Model 6 + $year^2$*vegetation type	6.23	0.0444	0.0324
3	Model 2 + year	22.62	0.0000	0.0000
2	Random intercept - fire	22.81	0.0000	0.0000
4	Model 3 + $year^2$	24.59	0.0000	0.0000
1	Null - fixed intercept	25.40	0.0000	0.0000
Native Perennial Grass Species (%)				
5	Model 4 + vegetation type	2.94	0.2294	0.1770
6	Model 5 + year*vegetation type	5.59	0.0611	0.0471
7	Model 6 + $year^2$*vegetation type	10.28	0.0059	0.0045
2	Random intercept - fire	25.66	0.0000	0.0000
3	Model 2 + year	27.52	0.0000	0.0000
1	Null - fixed intercept	27.60	0.0000	0.0000
4	Model 3 + $year^2$	28.63	0.0000	0.0000

Appendix 6-4. Parameters for four response variables for each of six herbaceous guilds in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to "natural regeneration" sites (creosote and thermic blackbrush vegetation). Percent is the variation (%) attributable to the random factor of Fire (Delamar, Duzak, Halfway, and Meadow Valley).

Source of variation	Parameter	S.E.	Z	P	Percent
Non-native Annual Grass					
Stem density (log)					
Constant	9.931	0.763	13.016	0.0000	
Year	-6.828	0.828	8.246	0.0000	
Year^2	1.551	0.201	7.716	0.0000	
Blackbrush	-3.605	0.980	3.679	0.0092	
NWPJ	-5.678	0.878	6.467	0.0000	
WPJ	-6.080	1.050	5.790	0.0000	
Year*Blackbrush	3.465	1.095	3.164	0.0008	
Year*NWPJ	5.784	0.970	5.963	0.0000	
Year*WPJ	5.634	1.170	4.815	0.0000	
Year^2*Blackbrush	-0.851	0.269	3.164	0.0008	
Year^2*NWPJ	-1.368	0.237	5.772	0.0000	
Year^2*WPJ	-1.235	0.287	4.303	0.0000	
Random effect					
Fire	0.018	0.019	0.947	0.1718	1.3
Non-native Annual Grass					
Stem density (%)					
Constant	2.182	0.215	10.149	0.0000	
Year	-1.31	0.233	5.622	0.0000	
Year^2	0.261	0.056	4.661	0.0000	
Blackbrush	-0.214	0.276	0.775	0.2192	
NWPJ	-0.981	0.247	3.972	0.0000	
WPJ	-1.233	0.296	4.166	0.0000	
Year*Blackbrush	0.130	0.308	0.422	0.3365	
Year*NWPJ	1.230	0.273	4.505	0.0000	
Year*WPJ	1.519	0.329	4.617	0.0000	
Year^2*Blackbrush	-0.048	0.076	0.632	0.2367	
Year^2*NWPJ	-0.266	0.067	3.970	0.0000	
Year^2*WPJ	-0.324	0.081	4.000	0.0000	
Random effect					
Fire	0.002	0.002	1.000	1	1.9

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Non-native Annual Grass Species (%)					
Constant	0.397	0.113	3.513	0.0000	
Year	0.305	0.122	2.500	0.0228	
$Year^2$	-0.106	0.029	3.655	0.0001	
Blackbrush	0.033	0.144	0.229	0.4094	
NWPJ	0.067	0.129	0.519	0.3019	
WPJ	0.133	0.154	0.864	0.1938	
Year*Blackbrush	0.135	0.161	0.839	0.2007	
Year*NWPJ	-0.202	0.143	1.413	0.0788	
Year*WPJ	-0.234	0.172	1.360	0.0869	
$Year^2$*Blackbrush	-0.051	0.040	1.275	0.1011	
$Year^2$*NWPJ	0.071	0.035	2.029	0.0212	
$Year^2$*WPJ	0.072	0.042	1.714	0.0432	
Random effect					
Fire	0.001	0.010	0.100	0.4602	3.3
Non-native Annual Forbs Stem density (log)					
Constant	2.117	0.623	3.398	0.0003	
Year	0.209	0.674	0.310	0.3783	
$Year^2$	0.109	0.163	0.669	0.2517	
Blackbrush	-3.482	0.799	4.358	0.0000	
NWPJ	-1.166	0.715	1.631	0.0514	
WPJ	-1.351	0.855	1.580	0.0571	
Year*Blackbrush	3.853	0.892	4.320	0.0000	
Year*NWPJ	-0.803	0.790	1.016	0.1548	
Year*WPJ	-0.960	0.953	1.007	0.1570	
$Year^2$*Blackbrush	-0.816	0.219	3.726	0.0001	
$Year^2$*NWPJ	0.114	0.193	0.591	0.2773	
$Year^2$*WPJ	0.147	0.233	0.631	0.2640	
Random effect					
Fire	0.021	0.019	1.105	0.1346	2.3

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Non-native Annual Forbs					
Stem density (%)					
Constant	-0.922	0.170	5.424	0.0000	
Year	1.573	0.183	8.596	0.0000	
Year^2	-0.340	0.044	7.727	0.0000	
Blackbrush	-0.072	0.217	0.332	0.3699	
NWPJ	1.064	0.194	5.485	0.0000	
WPJ	1.095	0.232	4.720	0.0000	
Year*Blackbrush	0.137	0.242	0.566	0.2857	
Year*NWPJ	-1.661	0.214	7.762	0.0000	
Year*WPJ	-1.770	0.258	6.860	0.0000	
Year^2*Blackbrush	-0.003	0.059	0.051	0.4797	
Year^2*NWPJ	0.389	0.052	7.481	0.0000	
Year^2*WPJ	0.409	0.063	6.492	0.0000	
Random effect					
Fire	0.003	0.002	1.500	0.0688	4.4
Non-native Annual Forbs					
Species (%)					
Constant	-0.175	0.122	1.434	0.0758	
Year	0.780	0.133	5.865	0.0000	
Year^2	-0.203	0.032	6.344	0.0000	
Blackbrush	0.043	0.158	0.272	0.3928	
NWPJ	0.359	0.142	2.528	0.0057	
WPJ	0.375	0.169	2.219	0.0132	
Year*Blackbrush	0.029	0.177	0.164	0.4349	
Year*NWPJ	-0.822	0.156	5.269	0.0000	
Year*WPJ	-0.914	0.189	4.836	0.0000	
Year^2*Blackbrush	-0.012	0.043	0.279	0.3901	
Year^2*NWPJ	0.230	0.038	6.053	0.0000	
Year^2*WPJ	0.251	0.046	5.457	0.0000	
Random effect					
Fire	0.001	0.010	0.100	0.4602	2.8

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Annual Forbs					
Stem density (log)					
Constant	4.432	0.474	9.350	0.0000	
Year	-4.759	0.517	9.205	0.0000	
Year^2	1.361	0.125	10.888	0.0000	
Blackbrush	-2.438	0.611	3.990	0.0000	
NWPJ	-2.868	0.548	5.234	0.0000	
WPJ	-2.903	0.656	4.425	0.0000	
Year*Blackbrush	2.164	0.684	3.164	0.0008	
Year*NWPJ	3.420	0.606	5.644	0.0000	
Year*WPJ	3.196	0.731	4.372	0.0000	
Year^2*Blackbrush	-0.542	0.168	3.226	0.0006	
Year^2*NWPJ	-0.987	0.148	6.669	0.0000	
Year^2*WPJ	-0.857	0.179	4.788	0.0000	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.2
Native Annual Forbs					
Stem density (%)					
Constant	0.294	0.060	4.900	0.0000	
Year	-0.310	0.054	5.741	0.0000	
Year^2	0.109	0.013	8.385	0.0000	
Blackbrush	0.006	0.052	0.115	0.4542	
NWPJ	0.133	0.047	2.830	0.0023	
WPJ	0.116	0.056	2.071	0.0192	
Year*Blackbrush	-0.051	0.023	2.217	0.0133	
Year*NWPJ	-0.069	0.021	3.286	0.0005	
Year*WPJ	-0.041	0.025	1.640	0.0505	
Random effect					
Fire	0.001	0.010	0.100	0.4602	3.2

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Annual Forbs					
Species					
Constant	8.333	1.018	8.186	0.0000	
Year	-9.293	1.104	8.418	0.0000	
Year^2	2.789	0.267	10.446	0.0000	
Blackbrush	-2.993	1.310	2.285	0.0112	
NWPJ	-4.376	1.176	3.721	0.0001	
WPJ	-4.360	1.404	3.105	0.0010	
Year*Blackbrush	2.906	1.460	1.990	0.0233	
Year*NWPJ	5.744	1.292	4.446	0.0000	
Year*WPJ	5.072	1.558	3.255	0.0006	
Year^2*Blackbrush	-0.848	0.358	2.369	0.0089	
Year^2*NWPJ	-1.738	0.315	5.517	0.0000	
Year^2*WPJ	-1.352	0.381	3.549	0.0002	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.1
Native Annual Forbs					
Species (%)					
Constant	1.462	0.164	8.915	0.0000	
Year	-1.270	0.179	7.095	0.0000	
Year^2	0.343	0.043	7.977	0.0000	
Blackbrush	-0.218	0.212	1.028	0.1520	
NWPJ	-0.627	0.190	3.300	0.0005	
WPJ	-0.840	0.227	3.700	0.0001	
Year*Blackbrush	-0.006	0.237	0.025	0.4900	
Year*NWPJ	0.717	0.210	3.414	0.0003	
Year*WPJ	0.846	0.253	3.344	0.0004	
Year^2*Blackbrush	0.019	0.058	0.328	0.3715	
Year^2*NWPJ	-0.192	0.051	3.765	0.0001	
Year^2*WPJ	-0.195	0.062	3.145	0.0008	
Random effect					
Fire	0.001	0.010	0.100	0.4602	1.6

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Perennial Forbs					
Stem density (log)					
Constant	2.014	0.418	4.818	0.0000	
Year	-2.091	0.456	4.586	0.0000	
$Year^2$	0.601	0.111	5.414	0.0000	
Blackbrush	-0.717	0.539	1.330	0.0918	
NWPJ	-2.495	0.484	5.155	0.0000	
WPJ	-1.694	0.578	2.931	0.0017	
Year*Blackbrush	0.380	0.603	0.630	0.2643	
Year*NWPJ	3.684	0.534	6.899	0.0000	
Year*WPJ	2.683	0.645	4.160	0.0000	
$Year^2$*Blackbrush	-0.039	0.148	0.264	0.3959	
$Year^2$*NWPJ	-0.985	0.131	7.519	0.0000	
$Year^2$*WPJ	-0.741	0.158	4.690	0.0000	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.2
Native Perennial Forbs					
Stem density (%)					
Constant	0.067	0.119	0.563	0.2867	
Year	-0.025	0.130	0.192	0.4239	
$Year^2$	0.017	0.031	0.548	0.2918	
Blackbrush	0.101	0.153	0.660	0.2546	
NWPJ	-0.278	0.137	2.029	0.0212	
WPJ	-0.112	0.164	0.683	0.2473	
Year*Blackbrush	-0.179	0.171	1.047	0.1475	
Year*NWPJ	0.535	0.152	3.520	0.0002	
Year*WPJ	0.406	0.183	2.219	0.0132	
$Year^2$*Blackbrush	0.048	0.042	1.143	0.1265	
$Year^2$*NWPJ	-0.137	0.037	3.703	0.0001	
$Year^2$*WPJ	-0.117	0.045	2.600	0.0047	
Random effect					
Fire	0.001	0.010	0.100	0.4602	2.9

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Perennial Forbs Species					
Constant	3.424	0.710	4.823	0.0000	
Year	-3.319	0.762	4.356	0.0000	
Year^2	1.017	0.184	5.527	0.0000	
Blackbrush	-1.291	0.912	1.416	0.0784	
NWPJ	-1.139	0.818	1.392	0.0820	
WPJ	-1.775	0.976	1.819	0.0345	
Year*Blackbrush	0.624	1.007	0.620	0.2676	
Year*NWPJ	3.280	0.890	3.685	0.0001	
Year*WPJ	4.161	1.072	3.882	0.0001	
Year^2*Blackbrush	-0.078	0.246	0.317	0.3756	
Year^2*NWPJ	-1.006	0.217	4.636	0.0000	
Year^2*WPJ	-1.180	0.262	4.504	0.0000	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.1
Native Perennial Forbs Species (%)					
Constant	0.429	0.152	2.822	0.0024	
Year	-0.200	0.166	1.205	0.1141	
Year^2	0.075	0.040	1.875	0.0304	
Blackbrush	-0.191	0.196	0.974	0.1650	
NWPJ	-0.060	0.176	0.341	0.3666	
WPJ	-0.038	0.210	0.181	0.4282	
Year*Blackbrush	-0.013	0.219	0.059	0.4765	
Year*NWPJ	0.513	0.194	2.644	0.0041	
Year*WPJ	0.526	0.234	2.248	0.0123	
Year^2*Blackbrush	0.030	0.054	0.556	0.2891	
Year^2*NWPJ	-0.161	0.047	3.426	0.0003	
Year^2*WPJ	-0.170	0.057	2.982	0.0014	
Random effect					
Fire	0.001	0.010	0.100	0.4602	1.9

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Annual Grass					
Stem density (log)					
Constant	2.005	0.473	4.239	0.0000	
Year	-1.541	0.516	2.986	0.0014	
$Year^2$	0.313	0.125	2.504	0.0061	
Blackbrush	0.918	0.610	1.505	0.0662	
NWPJ	-0.622	0.547	1.137	0.1278	
WPJ	-1.153	0.654	1.763	0.0390	
Year*Blackbrush	-1.004	0.683	1.470	0.0708	
Year*NWPJ	0.498	0.605	0.823	0.2053	
Year*WPJ	0.991	0.729	1.359	0.0871	
$Year^2$*Blackbrush	0.235	0.168	1.399	0.0809	
$Year^2$*NWPJ	-0.117	0.148	0.791	0.2145	
$Year^2$*WPJ	-0.221	0.179	1.235	0.1084	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.2
Native Annual Grass					
Stem density (%)					
Constant	0.402	0.045	8.933	0.0000	
Year	-0.309	0.051	6.059	0.0000	
$Year^2$	0.060	0.013	4.615	0.0000	
Random effect					
Fire	0.001	0.01	0.100	0.4602	3.1
Native Annual Grass					
Species					
Constant	1.540	0.266	5.789	0.0000	
Year	-1.311	0.290	4.521	0.0000	
$Year^2$	0.285	0.070	4.071	0.0000	
Blackbrush	0.160	0.343	0.466	0.3206	
NWPJ	-0.564	0.308	1.831	0.0336	
WPJ	-0.925	0.368	2.514	0.0060	
Year*Blackbrush	-0.214	0.384	0.557	0.2888	
Year*NWPJ	0.582	0.340	1.712	0.0434	
Year*WPJ	0.957	0.410	2.334	0.0098	
$Year^2$*Blackbrush	0.073	0.094	0.777	0.2186	
$Year^2$*NWPJ	-0.147	0.083	1.771	0.0383	
$Year^2$*WPJ	-0.219	0.100	2.190	0.0143	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.6

Appendix 6-4 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Annual Grass Species (%)					
Constant	0.576	0.100	5.760	0.0000	
Year	-0.478	0.109	4.385	0.0000	
$Year^2$	0.101	0.026	3.885	0.0001	
Blackbrush	0.121	0.129	0.938	0.1741	
NWPJ	-0.215	0.115	1.870	0.0307	
WPJ	-0.301	0.138	2.181	0.0146	
Year*Blackbrush	-0.123	0.144	0.854	0.1966	
Year*NWPJ	0.212	0.128	1.656	0.0489	
Year*WPJ	0.310	0.154	2.013	0.0221	
$Year^2$*Blackbrush	0.034	0.035	0.971	0.1658	
$Year^2$*NWPJ	-0.051	0.031	1.645	0.0500	
$Year^2$*WPJ	-0.071	0.038	1.868	0.0309	
Random effect					
Fire	0.001	0.010	0.100	0.4602	4.2
Native Perennial Grass Stem density (log)					
Constant	0.013	0.113	0.115	0.4542	
Year	0.026	0.100	0.260	0.3974	
$Year^2$	0.002	0.023	0.087	0.4653	
Blackbrush	0.047	0.097	0.485	0.3138	
NWPJ	0.217	0.088	2.466	0.0068	
WPJ	0.208	0.104	2.000	0.0228	
Year*Blackbrush	-0.029	0.043	0.674	0.2502	
Year*NWPJ	-0.077	0.039	1.974	0.0242	
Year*WPJ	-0.055	0.046	1.196	0.1158	
Random effect					
Fire	0.001	0.010	0.100	0.4602	1.0
Native Perennial Grass Stem density (%)					
Constant	0.006	0.023	0.261	0.3970	
Year	0.019	0.024	0.792	0.2142	
$Year^2$	-0.006	0.006	1.000	0.1587	
Blackbrush	0.002	0.010	0.200	0.4207	
NWPJ	0.033	0.008	4.125	0.0000	
WPJ	0.031	0.010	3.100	0.0010	
Random effect					
Fire	0.001	0.010	0.100	0.4602	1.4

Appendix 6-4 continued.

Source of variation	**Parameter**	**S.E.**	**Z**	**P**	**Percent**
Native Perennial Grass Species					
Constant	0.104	0.153	0.680	0.2483	
Year	0.070	0.165	0.424	0.3358	
$Year^2$	-0.008	0.041	0.195	0.4227	
Blackbrush	-0.064	0.062	1.032	0.1510	
NWPJ	0.186	0.055	3.382	0.0004	
WPJ	0.219	0.066	3.318	0.0005	
Random effect					
Fire	0.001	0.010	0.100	0.4602	0.3
Native Perennial Grass Species (%)					
Constant	0.023	0.049	0.469	0.3195	
Year	0.053	0.053	1.000	0.1587	
$Year^2$	-0.012	0.013	0.923	0.1780	
Blackbrush	-0.020	0.020	1.000	0.1587	
NWPJ	0.065	0.018	3.611	0.0002	
WPJ	0.063	0.021	3.000	0.0013	
Random effect					
Fire	0.001	0.01	0.100	0.4602	2.9

Appendix 6-5. Model selection statistics for six response variables for woody species in four areas (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to “natural regeneration” sites (creosote and thermic blackbrush vegetation). ΔAICc is the difference in the bias-corrected Akaike Information Criterion (AICc) between a given model and the best supported model, wAICc is the absolute support for a given model (= exp(-ΔAICc/2)), and rwAICc is the support relative to the other models. N_0 is overall species richness, N_1 is the exponentiation of Shannon’s index (H’), N_2 is the reciprocal of Simpson’s index of concentration (d), and $E_{1/d}$ is Simpson’s index of evenness.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
Stem density (log)					
	5	Model 4 + vegetation type	0.00	1.0000	0.8048
	6	Model 5 + year*vegetation type	2.96	0.2271	0.1828
	7	Model 6 + $year^2$*vegetation type	8.34	0.0154	0.0124
	1	Null - fixed intercept	230.84	0.0000	0.0000
	2	Random intercept - fire	232.85	0.0000	0.0000
	3	Model 2 + year	234.40	0.0000	0.0000
	4	Model 3 + $year^2$	235.64	0.0000	0.0000
Seedling density (log)					
	7	Model 6 + $year^2$*vegetation type	0.00	1.0000	0.9897
	6	Model 5 + year*vegetation type	9.14	0.0104	0.0103
	5	Model 4 + vegetation type	26.23	0.0000	0.0000
	4	Model 3 + $year^2$	45.75	0.0000	0.0000
	3	Model 2 + year	66.25	0.0000	0.0000
	1	Null - fixed intercept	147.16	0.0000	0.0000
	2	Random intercept - fire	149.17	0.0000	0.0000
N0					
	5	Model 4 + vegetation type	0.00	1.0000	0.5719
	6	Model 5 + year*vegetation type	0.85	0.6527	0.3733
	7	Model 6 + $year^2$*vegetation type	4.69	0.0958	0.0548
	3	Model 2 + year	220.86	0.0000	0.0000
	4	Model 3 + $year^2$	221.69	0.0000	0.0000
	1	Null - fixed intercept	241.13	0.0000	0.0000
	2	Random intercept - fire	243.14	0.0000	0.0000
N1					
	5	Model 4 + vegetation type	0.00	1.0000	0.8396
	6	Model 5 + year*vegetation type	3.54	0.1705	0.1431
	7	Model 6 + $year^2$*vegetation type	7.77	0.0205	0.0172
	3	Model 2 + year	161.14	0.0000	0.0000
	4	Model 3 + $year^2$	161.46	0.0000	0.0000
	1	Null - fixed intercept	177.64	0.0000	0.0000
	2	Random intercept - fire	179.65	0.0000	0.0000

Appendix 6-5 continued.

Model		**Variables**	**ΔAICc**	**wAICc**	**rwAICc**
N2					
	5	Model 4 + vegetation type	0.00	1.0000	0.8594
	6	Model 5 + year*vegetation type	3.82	0.1478	0.1270
	7	Model 6 + $year^2$*vegetation type	8.30	0.0158	0.0136
	3	Model 2 + year	129.77	0.0000	0.0000
	4	Model 3 + $year^2$	130.38	0.0000	0.0000
	1	Null - fixed intercept	143.75	0.0000	0.0000
	2	Random intercept - fire	145.77	0.0000	0.0000
Evenness[$E_{1/d}$]					
	5	Model 4 + vegetation type	0.00	1.0000	0.8828
	6	Model 5 + year*vegetation type	4.16	0.1250	0.1104
	7	Model 6 + $year^2$*vegetation type	9.74	0.0077	0.0068
	3	Model 2 + year	106.92	0.0000	0.0000
	1	Null - fixed intercept	107.37	0.0000	0.0000
	2	Random intercept - fire	108.01	0.0000	0.0000
	4	Model 3 + $year^2$	108.78	0.0000	0.0000

Appendix 6-6. Parameters for six response variables of woody species in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to "natural regeneration" sites (creosote and thermic blackbrush vegetation). Percent is the variation (%) attributable to the random factor of Fire (Delamar, Duzak, Halfway, and Meadow Valley). N_0 is overall species richness, N_1 is the exponentiation of Shannon's index (H'), N_2 is the reciprocal of Simpson's index of concentration (d), and $E_{1/d}$ is Simpson's index of evenness.

Source of variation	**Parameter**	**S.E.**	**Z**	**P**	**Percent**
Stem density (log)					
Constant	2.012	0.260	7.738	0.000	
Year	-0.119	0.273	-0.436	0.669	
$Year^2$	0.045	0.068	0.662	0.746	
Blackbrush	0.266	0.113	2.354	0.009	
NWPJ	1.285	0.095	13.526	0.000	
WPJ	1.231	0.112	10.991	0.000	
Random effect					
Fire	0.009	0.010	0.900	0.184	1.1
Seedling density (log)					
Constant	0.839	0.097	8.649	0.000	
Year	0.250	0.071	3.521	0.000	
$Year^2$	0.201	0.227	0.885	0.188	
Blackbrush	0.891	0.149	5.980	0.000	
NWPJ	0.289	0.127	2.276	0.011	
WPJ	0.026	0.141	0.184	0.427	
Year*Blackbrush	-0.449	0.099	-4.535	0.000	
Year*NWPJ	-0.170	0.087	-1.954	0.025	
Year*WPJ	-0.043	0.227	-0.189	0.425	
$Year^2$*Blackbrush	0.123	0.208	0.591	0.437	
$Year^2$*NWPJ	0.576	0.182	3.165	0.001	
$Year^2$*WPJ	0.583	0.218	2.674	0.004	
Random effect					
Fire	0.000	0.000	0.000	1.000	0.0

Appendix 6-6 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
N0					
Constant	1.111	1.206	0.921	0.179	
Year	0.563	1.286	0.438	0.331	
Year^2	-0.017	0.310	-0.055	0.478	
Blackbrush	0.433	1.547	0.280	0.390	
NWPJ	0.703	1.372	0.512	0.304	
WPJ	0.446	1.628	0.274	0.392	
Random effect					
Fire	0.094	0.082	1.146	0.126	3.0
N1					
Constant	0.741	0.343	2.160	0.015	
Year	0.832	0.357	2.331	0.010	
Year^2	-0.140	0.088	-1.591	0.056	
Blackbrush	-0.323	0.150	-2.153	0.016	
NWPJ	0.942	0.124	7.597	0.000	
WPJ	1.201	0.147	8.170	0.000	
Random effect					
Fire	0.024	0.024	1.000	1.000	1.6
N2					
Constant	0.836	0.290	2.883	0.002	
Year	0.637	0.304	2.095	0.018	
Year^2	-0.106	0.075	-1.413	0.079	
Blackbrush	-0.234	0.126	-1.857	0.032	
NWPJ	0.684	0.105	6.514	0.000	
WPJ	0.962	0.125	7.696	0.000	
Random effect					
Fire	0.011	0.012	0.917	1.000	1.0
$E_{1/d}$					
Constant	0.323	0.093	3.473	0.000	
Year	0.083	0.099	0.838	0.201	
Year^2	-0.013	0.024	-0.542	0.293	
Blackbrush	-0.047	0.040	-1.175	0.120	
NWPJ	0.233	0.034	6.853	0.000	
WPJ	0.264	0.040	6.600	0.000	
Random effect					
Fire	0.001	0.001	1.000	1.000	0.9

Appendix 6-7. Model selection statistics for absolute cover (%) for each of four herbaceous vegetation guilds in four areas (Delamar, Duzak, Halfway, and Meadow Valley) that burned in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to "natural regeneration" sites (creosote and thermic blackbrush vegetation). ΔAICc is the difference in the bias-corrected Akaike Information Criterion (AICc) between a given model and the best supported model, wAICc is the absolute support for a given model (= exp(-ΔAICc/2)), and rwAICc is the support relative to the other models.

Model		Variables	ΔAICc	wAICc	rwAICc
Non-native Annual Forbs					
	8	Model 7 + year^2*vegetation type	0.00	1.0000	0.5200
	7	Model 6 + year*vegetation type	0.16	0.9229	0.4800
	6	Model 5 + vegetation type	94.07	0.0000	0.0000
	5	Model 4 + year^2	467.05	0.0000	0.0000
	4	Model 3 + year	507.98	0.0000	0.0000
	3	Random intercept - fire	626.23	0.0000	0.0000
	2	Random intercept - plots	652.37	0.0000	0.0000
	1	Null - fixed intercept	859.87	0.0000	0.0000
Non-native Annual Grass					
	7	Model 6 + year*vegetation type	0.00	1.0000	0.5109
	8	Model 7 + year^2*vegetation type	0.09	0.9571	0.4890
	6	Model 5 + vegetation type	27.50	0.0000	0.0000
	5	Model 4 + year^2	44.59	0.0000	0.0000
	4	Model 3 + year	56.64	0.0000	0.0000
	3	Random intercept - fire	61.03	0.0000	0.0000
	2	Random intercept - plots	77.22	0.0000	0.0000
	1	Null - fixed intercept	183.42	0.0000	0.0000
Native Herbs					
	7	Model 6 + year*vegetation type	0.00	1.0000	0.8200
	8	Model 7 + year^2*vegetation type	4.22	0.1213	0.0994
	6	Model 5 + vegetation type	4.64	0.0982	0.0805
	5	Model 4 + year^2	181.49	0.0000	0.0000
	4	Model 3 + year	182.22	0.0000	0.0000
	3	Random intercept -fire	206.85	0.0000	0.0000
	2	Random intercept - plots	249.34	0.0000	0.0000
	1	Null - fixed intercept	340.82	0.0000	0.0000
Shrubs and Trees					
	6	Model 5 + vegetation type	0.00	1.0000	0.8911
	7	Model 6 + year*vegetation type	4.20	0.1222	0.1089
	5	Model 4 + year^2	82.12	0.0000	0.0000
	2	Random intercept - Plot	87.12	0.0000	0.0000
	4	Model 3 + year	87.28	0.0000	0.0000
	3	Random intercept - fire	89.14	0.0000	0.0000
	1	Null - fixed intercept	227.33	0.0000	0.0000

Appendix 6-8. Parameters for cover (%) of four herbaceous guilds in the Southern Nevada Complex fires of 2005. Mesic blackbrush (MBB), non-wilderness pinyon-juniper (NWPJ) and wilderness pinyon-juniper (WPJ) are vegetation types whose responses were coded relative to “natural regeneration” sites (creosote and thermic blackbrush vegetation). Percent is the variation (%) attributable to the random factors of Fire (Delamar, Duzak, Halfway, and Meadow Valley) and plot.

Source of variation	**Parameter**	**S.E.**	**Z**	**P**	**Percent**
Non-native Annual Forbs					
Fixed Factors					
Constant	0.262	0.033	7.939	0.000	
Year	-0.042	0.031	1.355	0.088	
$Year^2$	0.042	0.007	6.000	0.000	
NWPJ	-0.089	0.026	3.423	0.000	
WPJ	-0.131	0.030	4.367	0.000	
Blackbrush	0.137	0.029	4.724	0.000	
Year*NWPJ	-0.097	0.010	9.700	0.000	
Year*WPJ	-0.091	0.012	7.583	0.000	
Year*Blackbrush	-0.101	0.011	9.182	0.000	
Random Factors	**Parameter**	**S.E.**			
Fire	0.0001	0.0000			0.9
Plot	0.0050	0.0010			45.0
Error	0.0060	0.0000			54.1
Non-native Annual Grass					
Fixed Factors					
Constant	0.714	0.054	13.222	0.000	
Year	-0.260	0.051	5.098	0.000	
$Year^2$	0.050	0.012	4.167	0.000	
NWPJ	-0.168	0.042	4.000	0.000	
WPJ	-0.270	0.050	5.400	0.000	
Blackbrush	-0.148	0.047	3.149	0.001	
Year*NWPJ	0.064	0.016	4.000	0.000	
Year*WPJ	0.082	0.019	4.316	0.000	
Year*Blackbrush	0.006	0.018	0.333	0.370	
Random Factors	**Parameter**	**S.E.**			
Fire	0.0010	0.0010			3.4
Plot	0.0120	0.0020			41.4
Error	0.0160	0.0010			55.2

Appendix 6-8 continued.

Source of variation	Parameter	S.E.	Z	P	Percent
Native Herbs					
Fixed Factors					
Constant	0.237	0.094	2.521	0.006	
Year	-0.168	0.121	1.388	0.083	
Year^2	0.044	0.030	1.467	0.071	
NWPJ	-0.029	0.098	0.296	0.384	
WPJ	-0.001	0.109	0.009	0.496	
Blackbrush	-0.069	0.026	2.654	0.004	
Year*NWPJ	0.183	0.126	1.452	0.073	
Year*WPJ	0.146	0.139	1.050	0.147	
Year*Blackbrush	0.029	0.011	2.636	0.004	
Random Factors	**Parameter**	**S.E.**			
Fire	0.0001	0.0000			1.2
Plot	0.0020	0.0010			24.7
Error	0.0060	0.0000			74.1
Shrubs and Trees					
Fixed Factors					
Constant	0.013	0.015	0.867	0.193	
Year	0.031	0.016	1.938	0.026	
Year^2	-0.006	0.004	1.500	0.067	
NWPJ	0.045	0.009	5.000	0.000	
WPJ	0.034	0.010	3.400	0.000	
Blackbrush	-0.029	0.010	2.900	0.002	
Random Factors	**Parameter**	**S.E.**			
Fire	0.0001	0.0000			2.4
Plot	0.0020	0.0010			48.8
Error	0.0020	0.0010			48.8

Chapter 7: General Vegetation Trends and Seeded Species Establishment: A Descriptive Analysis Using Data From AA Macroplots

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INTRODUCTION

Monitoring of Emergency Stabilization and Rehabilitation (ESR) treatments is usually conducted for three growing seasons post-fire. Following the completion of this monitoring, land managers must make a variety of decisions regarding future land use including potential changes to livestock grazing. Land managers must also decide whether or not to implement the same ESR treatments in similar areas in the future, or if treatments should be altered, e.g. by changing rates and/or species used in seed mixes. Since monitoring is generally not conducted past this three-year time window, land managers must be able to make decisions in the face of uncertainty, using the best information available.

To aid them in making decisions about public lands that have experienced wildfire and subsequent ESR treatments, land managers often desire a spatial perspective describing what is occurring on the ground in treated areas in addition to a mechanistic understanding of why treatments may have succeeded or failed. These spatial perspectives help land managers separate out areas which may be reopened to normal land management practices from areas which may need more time for recovery.

In situations where monitoring funds are limited, and in large fire years where burned acreage is extensive, trade-offs must be made between more intensive monitoring strategies that provide a detailed examination of why ESR treatments succeed or fail and more extensive monitoring strategies that provide coarse information on what conditions are found on the ground over a large area. The Southern Nevada Complex (SNC) fires are an example in which large areas burned (597,096 acres) and large areas were aerially seeded (47,000 acres).

We adopted several monitoring strategies on the SNC fires that were intended to complement each other. We established more intensive paired seeded and unseeded demonstration plots with replicate subsamples (brushbelts) in each to evaluate the effects of seeding and not seeding. Analysis using these data is presented in Chapters 5, 6, and 8. We did not design these plots with the intention of covering the environmental heterogeneity present within the large seeding polygons, let alone larger fires. Therefore, we also established a more extensive monitoring strategy involving the use of Additional Aerial Seeding Coverage (AA) macroplots placed on transects to get better coverage of the large seeding polygons. The analysis presented in this chapter uses the data from the AA plots in order to provide land managers with a spatial perspective on what conditions were found within the seeding polygons over the first three growing seasons. This analysis highlights the importance of both place and scale in evaluating the effectiveness of ESR treatments.

METHODS

The analysis in this chapter uses data from the AA macroplots. These plots are located along transects and spaced 250 m apart. A more detailed description of the subsamples within an AA plot is found in Chapter 4.

Dominance Classes

To look at general vegetation trends across the seeding polygons, I used the ocular cover estimates from each AA plot to group the macroplot into a dominance class for each year it was sampled. I grouped each AA macroplot into one of nine dominance classes (Table 7-1). Each dominance class is based on a plant guild that commonly becomes dominant or co-dominant on the SNC. To be placed into a dominance class, the plant guild had to have 10% or more cover than the other guilds combined. If no plant guild had 10% or more cover than the other guilds combined, then I placed the macroplot into a mixed class. I then placed mixed class macroplots into sub-dominance classes based on co-dominance (Table 7-2). If the macroplot had less than 10% total live vegetation cover, then I placed it into a low vegetation cover class.

Table 7-1. Description of dominance classes used on the SNC fires. Dominance classes are defined using ocular cover data and are based upon certain guilds of plants that are commonly dominant on the SNC. To be in a class (other than low vegetation cover or mixed) the plant guild had to have 10% or more cover than the other guilds combined.

Dominance Class	**Description**
Non-Native Annual Grass	Macroplots dominated by non-native annual grasses. Common species include *Bromus tectorum* and *Bromus rubens*.
Erodium cicutarium	Macroplots dominated by non-native annual forb *Erodium cicutarium*.
Other Exotic Annual Forbs	Macroplots dominated by non-native annual forbs other than *Erodium*, including *Salsola tragus*, *Halogeton glomeratus*, *Sisymbrium altissimum* or a combination of these species.
Perennial Grasses	Macroplots dominated by perennial grass species, either native or exotic. Most common species include *Poa secunda*, *Elymus elymoides*, *Pleuraphis rigida*, and *Agropyron cristatum*.
Perennial Forbs	Macroplots dominated by perennial forb species (generally natives), including *Sphaeralcea ambigua*, *Penstemon* spp., *Heliomeris multiflora*, *Dichelostema capitatum*, and others.
Shrubs	Macroplots dominated by shrub species, often resprouters. Common species include *Quercus turbinella*, *Eriodictyon angustifolium*, *Purshia* spp., and *Amelanchier utahensis*.
Native Annual Forbs	Macroplots dominated by native annual forb species. Common species include *Gilia* spp., *Mentzelia albicaulis, Phacelia fremontii,* and others.
Low Vegetation Cover	Macroplots with less than 10% total live vegetation cover.
Mixed	Macroplots in which none of the above plant guilds have 10% or more cover than the other groups combined. Mixed plots are divided into sub-classes (Table 7-2).

Table 7-2. Description of sub-classes for mixed dominance macroplots in which two or more plant guilds co-dominate post-fire.

Mixed Sub-Class	Description
MEAP	Mixed Exotic Annuals and Perennials. Macroplots in which exotic annuals (e.g. *Bromus*, *Erodium*, *Salsola*) and perennials (grasses, forbs, and/or shrubs) co-dominate.
NAP	Native Annuals and Perennials. Macroplots in which native annual forbs and perennials co-dominate.
EAGEC	Exotic Annual Grasses and *Erodium cicutarium*. Macroplots in which non-native annual grasses (*Bromus tectorum* and/or *B. rubens*) and *Erodium cicutarium* co-dominate.
EAGS	Exotic Annual Grasses and Shrubs. Macroplots in which exotic annual grasses (*Bromus tectorum* and/or *B. rubens*) and shrub species co-dominate.
MP	Mixed Perennials. Macroplots in which a mix of perennials (shrubs, grasses, and/or forbs) co-dominate.
EANA	Exotic Annuals and Native Annual forbs. Macroplots in which non-native annuals (grasses or forbs) and native annual forbs co-dominate.

Seeded Species' Presence

Because seeded species are generally found only at low densities on the SNC, I looked at presence/absence of seeded species within each macroplot. I considered a seeded species present within the macroplot if it was recorded in either the density or ocular cover subsamples. Monitoring crews did not make a complete list of all species present in the AA macroplots in the ocular cover subsample, but they did pay special attention for any of the species used in the seed mixes within a macroplot and recorded their presence.

Spatial Distribution of Dominance Classes and Seeded Species

To look for spatial patterns in dominance and seeded species' presence, I mapped dominance classes and seeded species' presence/absence across the fires. These maps are found in the maps section in the Appendix A (Maps 11-33).

RESULTS

General Vegetation Trends

Table 7-3 summarizes the results for general vegetation trends based on data from the AA plots by seed mix.

Table 7-3. Number and percentages of plots in dominance classes and sub-dominance classes by seed mix and year. Data from sampled AA plots.

Mesic Blackbrush Seed Mix							
		2006		2007		2008	
Dominance Class	Mixed SubClass	N	%	N	%	N	%
Annual Grass		36	29.5	2	8	0	0
Erodium		4	3.3	1	4	92	85.2
Other Exotic Annuals		0	0	0	0	0	0
Perennial Grasses		0	0	0	0	0	0
Perennial Forbs		0	0	0	0	0	0
Shrubs		3	2.5	3	12	1	0.9
Native Annual Forbs		0	0	0	0	0	0
Low Vegetation Cover		25	20.5	6	24	6	5.6
Mixed	EAGEC	21	17.2	5	20	4	3.7
	EANA		0	0	0		0
	EAGS	17	13.9	1	4	1	0.9
	MEAP	16	13.1	7	28	4	3.7
	MP		0	0	0	0	0
	NAP		0	0	0	0	0
Total		122	100	25	100	108	100
Non-Wilderness PJ Seed Mix							
		2006		2007		2008	
Dominance Class	Mixed SubClass	N	%	N	%	N	%
Annual Grass		49	28.3	47	16.8	62	16.1
Erodium		0	0	2	0.7	5	1.3
Other Exotic Annuals		0	0	2	0.7	11	2.9
Perennial Grasses		0	0	4	1.4	2	0.5
Perennial Forbs		1	0.6	6	2.2	9	2.3
Shrubs		24	13.9	42	15.1	56	14.6
Native Annual Forbs		1	0.6	1	0.4	1	0.3
Low Vegetation Cover		42	24.3	61	21.9	24	6.3
Mixed	EAGEC	0	0	2	0.7	13	3.4
	EANA	5	2.9	1	0.4	3	0.8
	EAGS	26	15.0	40	14.3	39	10.2
	MEAP	14	8.1	41	14.7	125	32.6
	MP	5	2.9	28	10.0	28	7.3
	NAP	6	3.5	2	0.7	6	1.6
Total		173	100	279	100	384	100

Table 7-3. Continued.

Wilderness PJ Seed Mix							
		2006		2007		2008	
Dominance Class	Mixed SubClass	N	%	N	%	N	%
Annual Grass		15	24.2	5	7.0	20	13.3
Erodium		0	0	0	0	4	2.7
Other Exotic Annuals		0	0	0	0	2	1.3
Perennial Grasses		0	0			1	0.7
Perennial Forbs		1	1.6	1	1.4	3	2.0
Shrubs		13	21.0	13	18.3	20	13.3
Native Annual Forbs		0	0	4	5.6	0	0
Low Vegetation Cover		11	17.7	31	43.7	10	6.7
Mixed	EAGEC	0	0	0	0	4	2.7
	EANA	0	0	0	0	0	0
	EAGS	6	9.7	7	9.9	24	16.0
	MEAP	7	11.3	6	8.5	37	24.7
	MP	4	6.5	4	5.6	24	16.0
	NAP	5	8.1		0	1	0.7
Total		62	100	71	100	150	100

All Seed Mixes. In general across all seed mixes, the SNC is dominated by a mix of different plant guilds and also has many sites dominated by two or more plant guilds (Figure 7-1). In 2006, approximately 37% of sampled AA plots were considered mixed in which two or more plant groups co-dominated. Close to 28% of sampled AA plots were dominated by non-native annual grasses. Twenty-two percent of sampled AA plots had 10% or less total live vegetation cover and approximately 11% of AA plots were dominated by shrubs. Only a handful of other plots were dominated by other plant guilds including *Erodium cicutarium*, native annual forbs, and perennial forbs. No sites sampled were dominated by perennial grasses or non-native annual forbs other than *Erodium*.

In 2007, conditions were fairly similar to 2006 across all seed mix AA macroplots. The number of mixed dominance class sites stayed essentially the same with 38% of all sampled AA macroplots. Most notable was a 50% decline in the percentage of plots dominated by annual grasses alone (from 28% of all sampled plots to 14%). This decline was coupled with small increases in shrub-dominated sites (from 11% to 15.5%) and in sites with low total vegetation cover (from 22% to 26%). There were also increases in the handful of sites dominated by other plant guilds, including some sites in 2007 dominated by perennial grasses, other non-native annual forbs (i.e. *Salsola tragus*), and perennial forbs.

In 2008, in all AA plots sampled across all seed mixes, there were some notable changes. Sites dominated by non-native annual grasses remained at a similar percentage of total sampled plots (13%). There was an increase of mixed dominance class sites to

approximately 49% of the total sites. There was also a large increase in the percentage of plots dominated by *Erodium* to close to 16% of all sampled plots. Alternately, there was a strong decrease in sites with low vegetation cover to approximately 6% of all sampled plots.

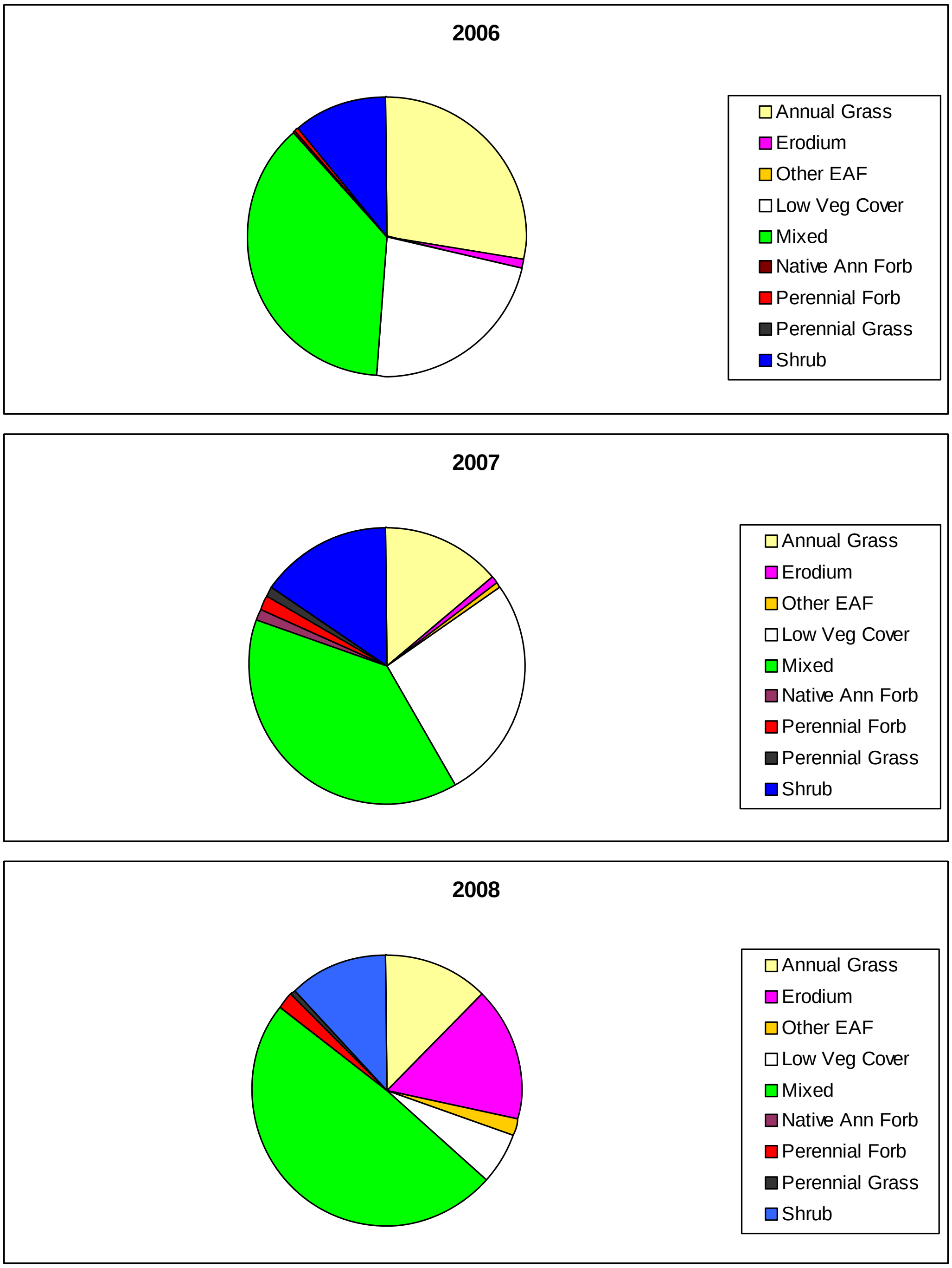

Figure 7-1. Dominance classes for all seed mixes from 2006 to 2008. The mixed class is placed into sub-classes in Figure 7-2. Other EAF are exotic annual forbs other than *Erodium cicutarium* (e.g. *Salsola tragus*).

Shrub-dominated sites decreased slightly to approximately 12% of all sampled plots. There was also an increase in the percentage of plots dominated by exotic annual forbs other than *Erodium*. These plots were generally dominated by Russian thistle (*Salsola tragus*). Only a handful of sites were dominated by other plant guilds (perennial grasses, perennial forbs, native annual forbs).

Mixed Dominance Classes. Mixed sites in which two or more plant guilds co-dominated make up the largest percentage of total AA plots sampled for all three years. In 2008, mixed dominance class sites represented close to 50% of all of the AA plots sampled. Therefore, it is important to look more in depth at which plant guilds make up the major components of these mixed sites. Across all three years of sampling, mixed sites generally include perennials as a co-dominant plant group (Figure 7-2). Mixed sites often include an exotic annual species (usually either *Bromus* or *Erodium*) as a co-dominant as well. Mixed Exotic Annual/Perennial (MEAP)-dominated plots represent the largest percentage of mixed sites in 2007 and 2008 at 37% and 53% respectively. These sites contain a mix of *Bromus* grasses and/or *Erodium* as well as perennials including grasses, forbs, and/or shrubs. These sites do not include those that are co-dominated specifically by annual grasses and shrubs, which represent a large portion of the mixed sites. Non-native annual grass and shrub co-dominated sites (EAGS) represented the largest percentage of mixed sites in 2006 with 38%, and the second most in 2007 and 2008 with 33% and 20% of total mixed sites respectively. Sites co-dominated by a mix of perennials only (MP) were also quite common, especially in 2007 and 2008 when they represented 22% and 17% of the total mixed sites respectively. There was an increase in percentage of mixed site AA plots dominated by only mixed perennials from 2006 to 2007. Sites dominated by a mix of exotic annual grasses and *Erodium* only (EAGEC) represented a rather small percentage of the mixed sites and declined from 2006 to 2008 (16% of sites in 2006, 5% in 2007, and 7% in 2008). Mixed sites that were the least common were those in which native annuals were a co-dominant either mixed with exotic annuals (EANA) or perennials (NAP).

Mesic Blackbrush Seed Mix. In 2006, 44% of the sampled AA plots in the Mesic Blackbrush Seed Mix polygons were dominated by two or more plant guilds. Close to 30% of the sampled plots were dominated by non-native annual grasses. Slightly more than 20% of the sites had low vegetation cover. Approximately 2% of the plots were dominated by shrubs and 3% of the plots were dominated by *Erodium cicutarium* (Figure 7-3). Most of the mixed sites contained a shrub component and a non-native annual component, consisting of

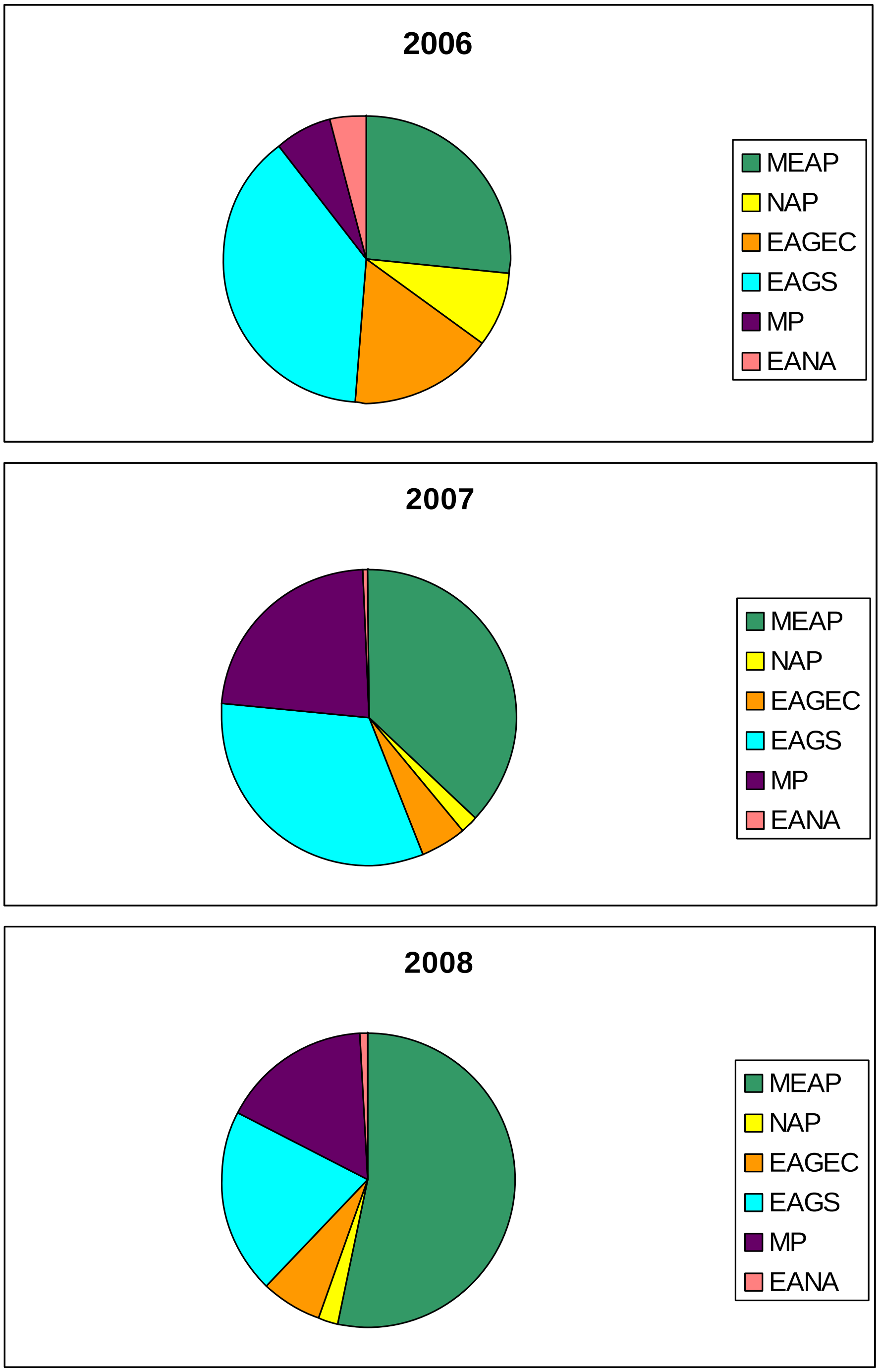

Figure 7-2. Mixed dominance class sub-classes. MEAP = Mixed Exotic Annuals and Perennials; NAP = Native Annuals and Perennials; EAGEC = Exotic Annual Grasses and *Erodium cicutarium*; EAGS = Exotic Annual Grasses and Shrubs; MP = Mixed Perennials; EANA = Exotic Annuals and Native Annuals.

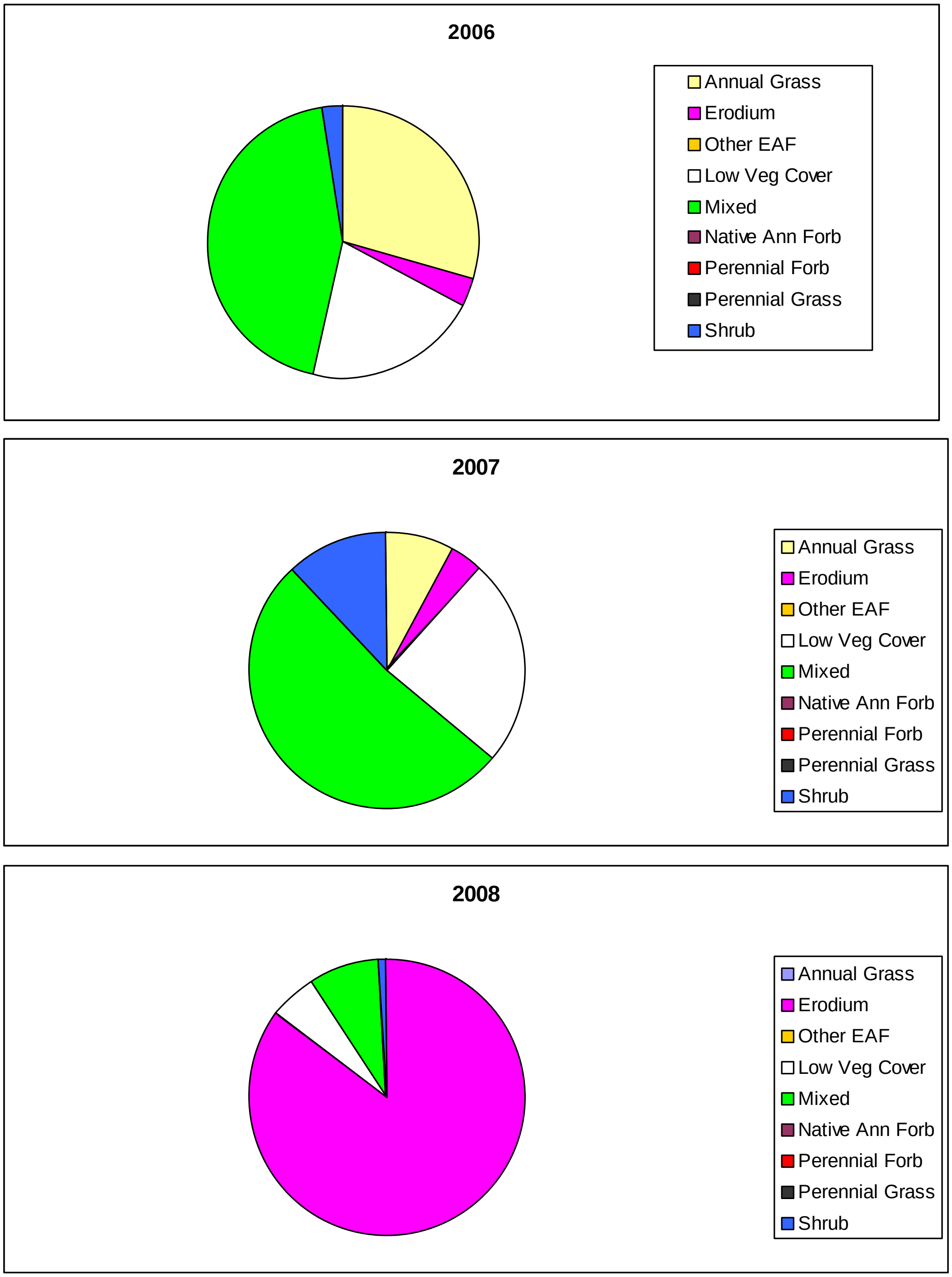

Figure 7-3. Dominance classes in the Mesic Blackbrush Seed Mix macroplots.

either annual grasses, *Erodium cicutarium* or both. Of the mixed sites 31.5% were dominated by a mix of annual grasses and shrubs (EAGS); close to 30% were considered MEAP sites containing shrubs and annual grasses/*Erodium*, and 39% were co-dominated by non-native annual grasses and *Erodium* (EAGEC).

In 2007, crews sampled only 25 AA plots in the Mesic Blackbrush Seed Mix polygons. This is a low sample size and makes the results somewhat suspect. Of these 25 AA plots, 52% were classified as mixed dominance sites. Only 8% were dominated by annual grasses. Four percent of the plots were dominated by *Erodium*, and 12% of the plots were dominated by shrubs. In 2007, the mixed dominance sites generally were dominated by both annual grasses and *Erodium* (42% EAGEC) or by a mix of perennials and non-native annuals (58% MEAP generally *Erodium* and shrubs).

In 2008, there was a significantly large shift to sites dominated by *Erodium cicutarium*, to approximately 85% of the sampled plots. No plots were dominated by annual grasses alone. Approximately 6% of the sites had low vegetation cover and 8% of the sites were dominated by two or more plant guilds. Less than 1% of the sites were dominated by shrubs. In 2008, the mixed sites were co-dominated by non-native annual grasses and *Erodium* (44% EAGEC), a mix of perennials and non-native annuals (44% MEAP generally *Erodium* and shrubs), and annual grasses and shrubs (11% EAGS).

Non-Wilderness Pinyon-Juniper (PJ) Seed Mix. In 2006, 28% of the sampled Non-Wilderness PJ Seed Mix AA plots were dominated by annual grasses (Figure 7-4). Twenty-four percent of the plots had low vegetation cover. Close to 33% of the plots were co-dominated by two or more plant guilds. Close to 14% of the sampled AA plots were shrub-dominated. Less than 1% of the plots were dominated by native annual forbs, and less than 1% of plots were dominated by perennial forbs. Of the plots co-dominated by two or more plant guilds, most (71.4%) contained both a perennial and non-native annual component. Of these, roughly two-thirds were dominated by a mix of annual grasses and shrubs (EAGS) and many of the others were dominated by a mix of annual grasses and perennial forbs (MEAP sites). Other sites containing two or more plant groups co-dominating include exotic annuals and native annuals (9% EANA), native annuals and perennials (11% NAP), and perennials only (9% MP).

In 2007, the percentage of sampled AA plots dominated by annual grasses decreased from 28% to 17%. The percentage of AA plots co-dominated by two or more plant guilds increased to 41%. Shrub-dominated sites remained roughly the same as in 2006 at 15% of sampled plots. A handful of other dominance classes were occasionally found in 2007 including perennial grasses (1.4%), perennial forbs (2.2%), *Erodium* (0.7%), other exotic annual forbs (0.4%) and native annual forbs (0.4%). Of the 41% of the sites dominated by two or more plant guilds, most (71.4%) were dominated by a mix of exotic annuals and

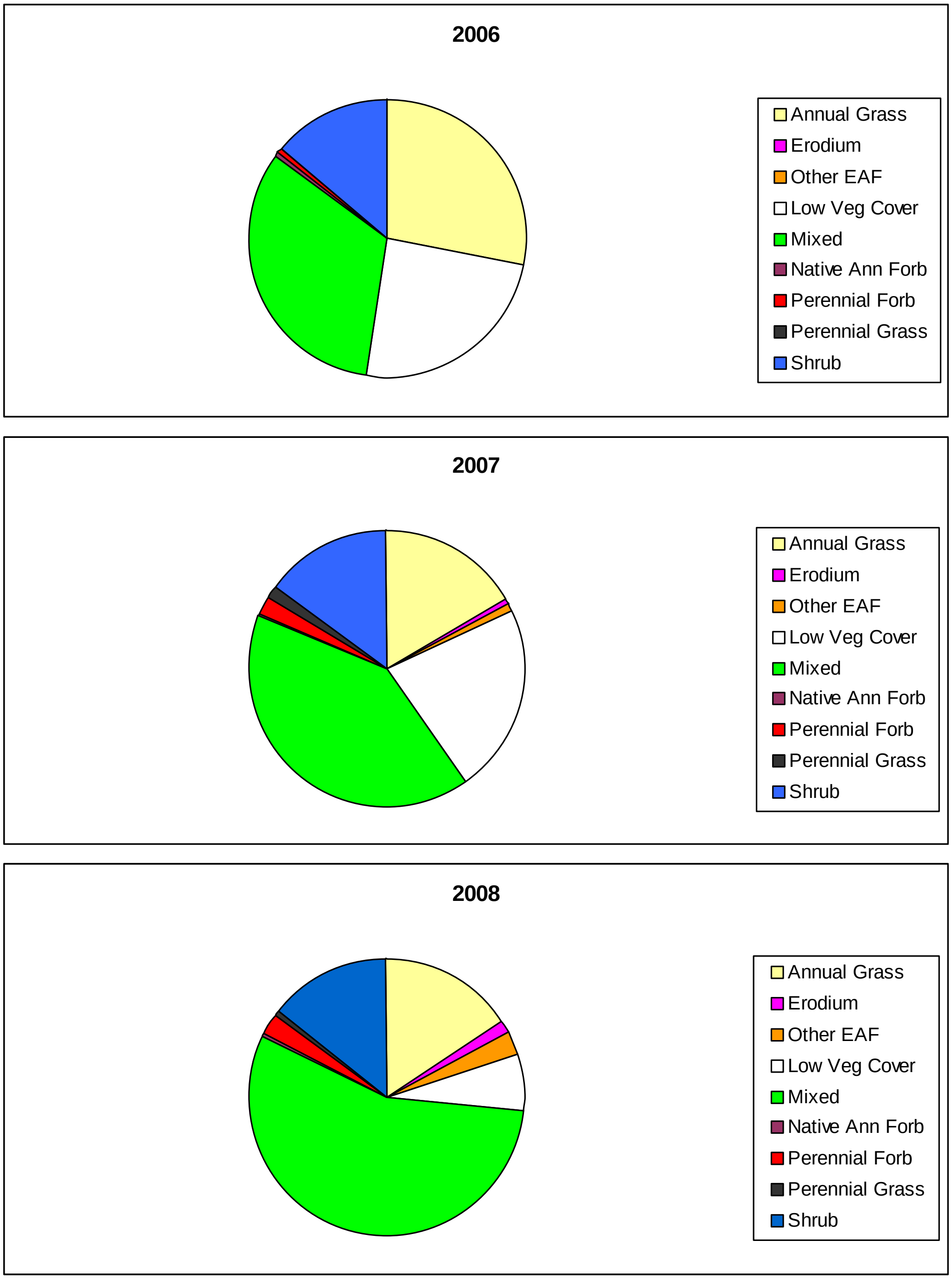

Figure 7-4. Dominance classes for the Non-Wilderness PJ Seed Mix AA macroplots.

perennials, including annual grasses and shrubs (EAGS), and annual grasses and other perennials (MEAP). Approximately 25% of the mixed dominance sites were co-dominated by a mix of perennials (MP). A handful of other mixed dominance sites were found, including native annuals and perennials (2% NAP), and exotic annuals and native annuals (1% EANA). Less than 2% of the mixed sites were dominated only by exotic annuals.

In 2008, the percentage of AA plots dominated by annual grasses remained roughly the same as in 2007 at 16% of sampled plots. Sites with low total vegetation cover decreased to approximately 6% of sampled plots. Sites co-dominated by two or more plant guilds increased to 55% of sampled plots. A few sites dominated by other plant guilds occurred in the Non-Wilderness PJ Seed Mix polygons, including: *Erodium*-dominated sites (1.3%); sites dominated by other exotic annual forbs (generally *Salsola tragus* (2.9%); perennial grass-dominated sites (0.5%); perennial forbs (2.3%); and native annual forbs (0.3%). Of the sites co-dominated by two or more plant guilds, most (76.6%) were dominated by a mix of non-native annuals and perennials. Thirteen percent were dominated by a mix of perennials only (MP). Over 6% of the sampled plots were dominated by a mix of exotic annuals including grasses, *Erodium*, and/or *Salsola tragus*. In 2008, *Salsola tragus* became more common in the Non-Wilderness PJ polygons, and was dominant in a few areas. Of the mixed dominance sites, 2.8% were dominated by native annuals and perennials (NAP).

Wilderness PJ Seed Mix. In 2006, roughly 25% of the sampled AA plots in the Wilderness PJ Seed Mix were dominated by non-native annual grasses (Figure 7-5). Thirty-four percent of sites were co-dominated by two or more plant guilds. Eighteen percent of the sampled plots had low total vegetation cover. More than 21% of sites were dominated by shrubs. Less than 2% of sites were dominated by perennial forbs. Of the 34% of sites co-dominated by two of more plant guilds, close to 60% were dominated by a mix of perennials and non-native exotic annuals, especially a mix of annual grasses and shrubs. Twenty-three percent of these mixed sites were dominated by native annuals and perennials and 18% were dominated by mixed perennial species only.

In 2007, only 7% of the sampled AA plots in the Wilderness PJ Seed Mix were dominated by non-native annual grasses. There was a large increase in percentage of sampled plots with low total vegetation cover, to 43.4% of sampled plots. Twenty-four percent of sampled plots were co-dominated by two or more plant guilds, and roughly 18% of sampled plots were dominated by shrubs. Close to 6% of sites were dominated by native annual forbs and 1.4% of sites were dominated by perennial forbs. Of the 24% of sites dominated by two or more plant guilds, most (76.6%) were co-dominated by perennials and exotic annuals. The remaining plots were dominated by a mix of perennials only (MP).

In 2008, 13.3% of the sampled AA plots in the Wilderness PJ Seed Mix were dominated by non-native annual grasses. There was a strong decrease in sites with low total

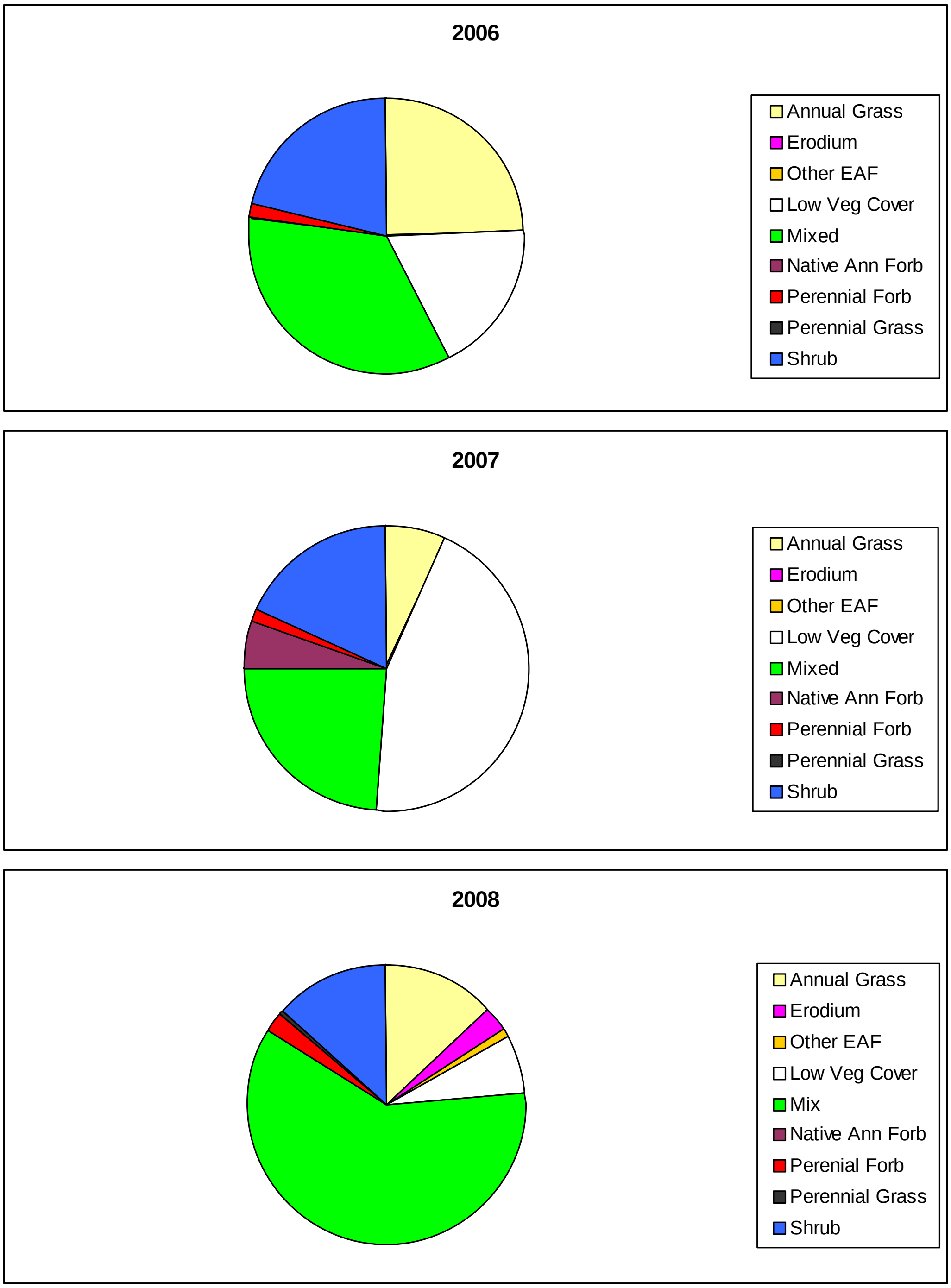

Figure 7-5. Dominance classes for the Wilderness PJ AA macroplots.

vegetation cover to approximately 7% of sampled plots. A majority of sampled plots (60%) had two or more groups of plants co-dominant. Of the plots sampled 13.3% were dominated by shrubs; 2.7% were dominated by *Erodium*; and 1.3% were dominated by *Salsola tragus*. Two percent were dominated by perennial forbs and less than 1% was dominated by perennial grasses. Of the 60% of the sampled plots dominated by two or more plant guilds, a majority (67.8%) was dominated by mixed perennials and exotic annuals (MEAP and

EAGS). Close to 27% were dominated by mixed perennials only (MP), and 4.4% were dominated by exotic annuals only. Approximately 1% of these mixed dominance class plots were native annuals/perennials (NAP).

Seeded Species' Presence/Absence

In general, seeded species are not found in high abundance within the SNC aerial seeding polygons. With the exception of a few localized areas in which crested wheatgrass (*Agropyron cristatum*) was found at high densities in the Clover Mountains portion of the Non-Wilderness PJ Seed Mix, seeded species never achieved dominance or co-dominance in the areas in which they were seeded. However, seeded species increased in presence over the three-year period across all three seed mixes (Figure 7-6).

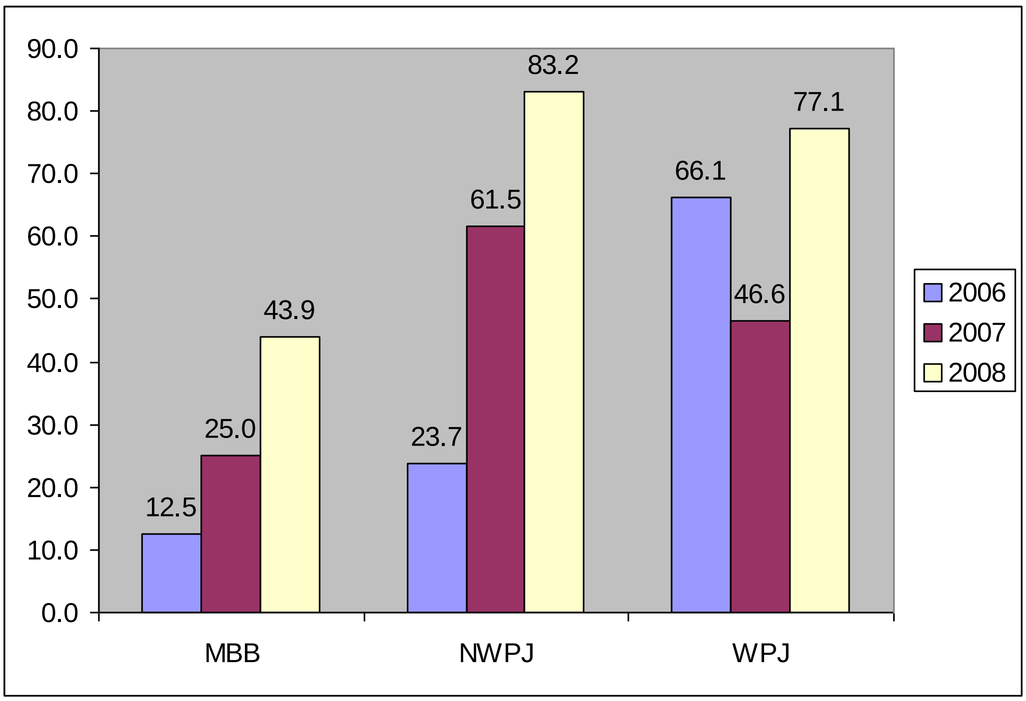

Figure 7-6. Percentage of sampled AA macroplots with seeded species present within the Mesic Blackbrush Seed Mix (MBB), Non-Wilderness PJ Seed Mix (NWPJ), and Wilderness PJ Seed Mix (WPJ), 2006-2008.

Mesic Blackbrush Seed Mix. The Mesic Blackbrush Seed Mix had the fewest seeded species present of the three seed mixes across all three years (Figure 7-6). However, seeded species' presence increased from 12.5% of plots in 2006 to close to 44% of plots in 2008. Data from AA macroplots show an increase in presence of most species over the three-year period (Table 7-4). This is especially true with forage kochia (*Bassia prostrata*), which was found in close to 15% of the sampled AA plots in 2008 and sand dropseed (*Sporobolus cryptandrus*), which was found in close to 16% of the sampled AA plots. Reconnaissance in June 2009 revealed that seeded species appear to becoming even more abundant in the mesic

blackbrush seeding polygons. This is especially true of small burnet (*Sanguisorba minor*), forage kochia (*Bassia prostrata*), and fourwing saltbush (*Atriplex canescens*). Two species (*Grayia spinosa* and *Elymus elymoides*) used in the seed mix were never seen in any of the AA macroplots.

Table 7-4. Percentage of sampled AA plots in the Mesic Blackbrush Seed Mix polygons containing a seeded species used in the mix.

Species	2006	2007	2008
Achnatherum hymenoides (Indian ricegrass)	5.7	0	8.3
Atriplex canescens (Fourwing saltbush)	0	0	4.6
Bassia prostrata (Forage kochia)	0	0	14.8
Elymus elymoides (Bottlebrush squirreltail)	0	0	0
Grayia spinosa (Spiny hopsage)	0	0	0
Linum perenne (Blue flax)	3.3	4.0	8.3
Penstemon palmeri (Palmer's penstemon)	0	0	1.9
Poa secunda (Sandberg bluegrass)	0	0	1.9
Sanguisorba minor (Small burnet)	1.7	20.0	2.8
Sporobolus cryptandrus (Sand dropseed)	0	0	15.7
Pleuraphis jamesii (Galleta grass)	0	0	5.6

Non-Wilderness PJ Seed Mix. Following the 2008 growing season, the Non-Wilderness PJ Seed Mix had the highest percentage of plots with seeded species present at 83.2% (Figure 7-6). It also had the largest increase in presence of seeded species over the three-year monitoring period. The most commonly present seeded species were bottlebrush squirreltail (*Elymus elymoides*) at 48.8% and crested wheatgrass (*Agropyron cristatum*) at 47.4% in 2008 (Figure 7-7; Table 7-5). Palmer's penstemon (*Penstemon palmeri*) and Sandberg bluegrass (*Poa secunda*) were found present in more than 30% of the sampled AA plots in the Non-Wilderness PJ Seed Mix polygons. One species (*Elymus wawawaiensis*) was never seen in any of the macroplots. The data generally show an increase in presence of each seeded species over the three-year period with the exception of two species (*Agropyron fragile* and *Elymus lanceolatus*) which declined in abundance. In the case of Siberian wheatgrass, it is very difficult to distinguish from crested wheatgrass, and individuals of *A. fragile* may have been misidentified as *A. cristatum*.

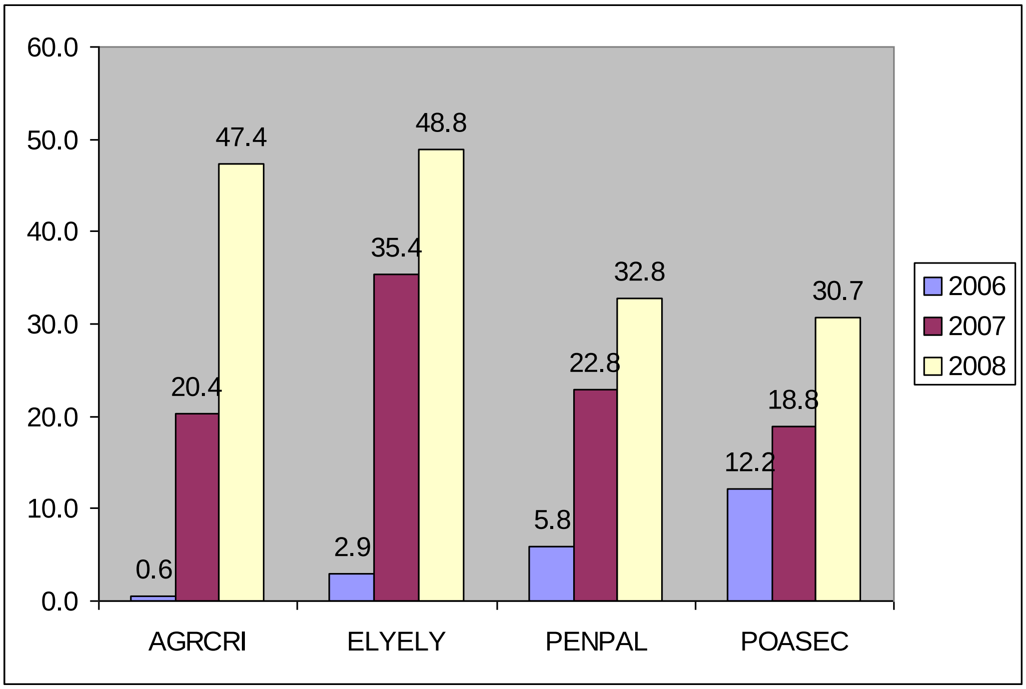

Figure 7-7. Increases in percent present of four commonly occurring seeded species in the Non-Wilderness PJ Seed Mix polygons, 2006-2008. AGRCRI = *Agropyron cristatum*; ELYELY = *Elymus elymoides*; PENPAL = *Penstemon palmeri*; POASEC = *Poa secunda*.

Table 7-5. Percentage of sampled AA plots in the Non-Wilderness PJ Seed Mix polygons containing a seeded species used in the mix.

Species	2006	2007	2008
Achnatherum hymenoides (Indian ricegrass)	5.8	6.2	11.9
Agropyron cristatum (Crested wheatgrass)	0.6	20.4	47.4
Agropyron fragile (Siberian wheatgrass)	0.6	0.4	0
Elymus elymoides (Bottlebrush squirreltail)	2.9	35.4	48.8
Elymus lanceolatus (Thickspike wheatgrass)	1.7	1.1	0.3
Elymus wawawaiensis (Snake River wheatgrass)	0	0	0
Penstemon palmeri (Palmer’s penstemon)	5.8	22.8	32.8
Poa secunda (Sandberg’s bluegrass)	12.2	18.8	30.7

Wilderness PJ Seed Mix. Data from the AA macroplots generally show a decline in seeded species’ presence in 2007 and then an increase in 2008. The decline in 2007 is coupled with a high proportion of plots having low vegetation cover (Figure 7-5). All species used in the Wilderness PJ Seed Mix were found within sampled plots (Table 7-6). They are also all native species that are commonly found in the plant communities in which they were seeded. Bottlebrush squirreltail (*Elymus elymoides*) is the most commonly occurring seeded species and was found in 53.6% of sampled AA plots in 2008. Crested wheatgrass was not seeded in the Wilderness PJ Seed Mix but was found in two AA macroplots in the Wilderness PJ Seed Mix polygons. Both of these plots were within 500m of the Non-Wilderness PJ Seed Mix, where crested wheatgrass was seeded.

Table 7-6. Percentage of sampled AA plots in the Wilderness PJ Seed Mix polygons containing a seeded species used in the mix.

Species	2006	2007	2008
Achnatherum hymenoides (Indian ricegrass)	7.6	7.0	21.7
Elymus elymoides (Bottlebrush squirreltail)	10.6	33.3	53.6
Penstemon palmeri (Palmer's penstemon)	17.7	2.7	36.7
Poa secunda (Sandberg bluegrass)	53.2	13.9	37.7
Hesperostipa comata (Needleandthread grass)	12.9	2.8	19.1

DISCUSSION

Seeded Species Establishment

Seeded species are generally only establishing at very low densities on the SNC (see Chapters 5-6). After three growing seasons, seeded species have not had ecologically meaningful effects. They have not provided meaningful competition against non-native annual grasses or *Erodium*. However, their presence in the seeding polygons in which they were seeded has increased over the three-year monitoring period. In 2008, seeded species were found growing in more than three-quarters of sampled AA plots in both of the PJ seed mixes and in nearly half of the sampled plots in the Mesic Blackbrush Seed Mix.

Land managers typically believe that seeding treatments can take more than three growing seasons to establish. It is possible that this could be true with the SNC aerial seeding treatments. We have observed an increase in the presence of seeded species in all seed mixes on the SNC. Reconnaissance in 2009—four growing seasons after treatment—revealed potentially higher levels of seeded species establishment in the Mesic Blackbrush Seed Mix polygons. This was especially true of forage kochia—a species whose seed supposedly loses viability rapidly. Because seeded species appear to be increasing in the SNC seeding polygons, these treatments should not yet be written off as failures.

Many of the seeded species used in the mixes on the SNC are native perennials that commonly regenerate naturally post-fire in the regions that were seeded. From our data, it is hard to separate which individuals of a given species established from seed from the treatment and which established from seed found naturally in the ecosystem. Other sampling designs used for this monitoring project (see Chapters 4-6) involved the comparison of paired seeded and unseeded control areas. However, it is difficult to be certain that these controls did not in fact receive seed. Light seeds are likely to blow long distances in the wind. (Some are specifically designed to do just this.) For an airplane to seed around a designated control it must fly on all sides of the control, so no matter in which direction the wind is blowing at some point during the seeding treatment seed will be blowing in the direction of the control.

Sampling points within the control plots were often less than 100 m away from directly below where the airplane depositing seed was flying.

One of the more commonly establishing species seeded in all three mixes is bottlebrush squirreltail (*Elymus elymoides).* In 2008, it was found present in approximately half of all the Non-Wilderness and Wilderness PJ Seed Mix AA plots (Tables 7-4, 7-5, and 7-6). It was never sighted in the Mesic Blackbrush Mix AA plots. Maps 19-21 (Appendix A) show the increase in its distribution across the PJ seed mixes. This native species commonly establishes post-fire in the region and may or may not be establishing as a result of the seeding treatment.

A handful of other species used in the Non-Wilderness PJ and Mesic Blackbrush seed mixes are non-natives. These species would not typically be found in the region unless they were used in historical land treatments. Since the Wilderness and Non-Wilderness PJ seed mix polygons are in similar areas ecologically, this allows for a good comparison for non-native species used in the mixes.

One of the more commonly establishing seeded non-native grass species is crested wheatgrass (*Agropyron cristatum*). This species was only seeded in the Non-Wilderness PJ Seed Mix. In 2008, it was found in nearly half of the sampled AA plots in the Non-Wilderness PJ Seed Mix. Maps 16-18 (Appendix A) show its increase in distribution across all of the Non-Wilderness PJ Seed Mix polygons. Prior to 2008, it was found only in the Non-Wilderness PJ Seed Mix. In 2008, it was found in only two plots outside of the area in which it was seeded. Both of these plots were located in the Wilderness PJ Seed Mix and were within 500m of the Non-Wilderness PJ Seed Mix. It has been argued that crested wheatgrass is establishing in the SNC as a result of previous seeding treatments. However, there is no specific knowledge of it being seeded in the areas seeded as part of the SNC. Furthermore, if it were establishing as a result of an historic land use treatment, one would expect it to be found more frequently farther away from the Non-Wilderness PJ Seed Mix (more often in the Wilderness PJ Seed Mix for example). If it were establishing from a previous seeding, one would also expect it to be found not uniformly throughout the Non-Wilderness Seed Mix polygons but rather only in a portion of the seeding area. However, crested wheatgrass is found throughout the entire region in which it was seeded and nearly nowhere else (Map 18, Appendix A). Therefore, I conclude that it is very probable that the crested wheatgrass establishing in the SNC seeding polygons is due to the SNC seeding treatment.

General Vegetation Trends

Results from other chapters show that non-native annual grasses and *Erodium* are found at much higher mean densities than native species on the SNC. However, individuals of these

species tend to be smaller than native perennial species. While there may be many more individuals of *Bromus* and *Erodium* than native perennials on the SNC, it is not fair to say that these non-natives dominate the landscape based on density data alone. For instance, there are likely to be many more individuals of Sandberg bluegrass in a dense pinyon-juniper woodland than there are pinyon and juniper trees. However, it would be a mistake to conclude that the area was dominated by Sandberg bluegrass rather than pinyon and/or juniper, because the trees are so much larger than the individual grass plants. The results presented in this chapter are based on cover data from the AA plots. Percent cover is a better indicator of dominance than density. Cover better accounts for the size of individuals.

The results from this chapter reveal a landscape dominated by a mix of different types of plants—both perennials and annuals, both natives and non-natives. The data from this chapter also draw a fairly strong distinction in plant dominance between burned higher elevation pinyon-juniper woodlands and burned mesic blackbrush communities. At the higher elevations, a mix of different plant guilds dominate and a mix of different plant guilds often co-dominate. In fact, mixed dominance in which two or more plant guilds co-dominate were found in over half of the sampled AA plots in both PJ seed mixes in 2008. Most of these mixed sites contained a type of perennial as a co-dominant. In many cases, this was due to resprouting shrub species that can be considered an interior chaparral component (e.g. *Quercus turbinella*, *Garrya flavescens*, *Amelanchier utahensis*, etc.). It appears that these higher elevation areas may have some resiliency to fire.

At the lower elevations, from sites in the Mesic Blackbrush Seed Mix, we see a landscape dominated primarily by *Erodium* in 2008. This was a shift from 2006, in which many sites were dominated by annual grasses or two or more plant guilds. (This appears to be precipitation related and was reported regionwide.) Nevertheless, sites at these lower elevations did tend to have a perennial component. Resprouting shrubs such as *Yucca baccata* and *Purshia glandulosa* were also common. In some areas perennial grasses such as *Aristida purpurea* were returning in high abundance.

Another trend that is apparent from the results presented in this chapter is that annual grass dominance has declined over the three-year period. In the PJ seed mix polygons, there has been a shift from areas dominated by annual grasses alone to areas dominated by a mix of perennials and exotic annuals. In the lower elevations, this shift has been more from annual grasses to *Erodium* dominance.

Lastly, another lesson from the results presented in this chapter is that post-fire vegetation communities in the Great Basin-Mojave Transition Zone can change rapidly in the first few growing seasons post-fire. Changes are not always uniform across the entire burned region but rather can vary within the region. For example, across much of the SNC we saw

an increase in sites with less than 10% vegetation cover in the second growing season. This was most pronounced in the Wilderness PJ Seed Mix polygons and may have resulted from a crop of annual grasses not returning in the second growing season due to moisture conditions. In the western Non-Wilderness PJ seeding polygon on the Delamar Fire, however, the opposite pattern occurred. Annual grasses increased there in 2007. Localized seasonal variability in moisture conditions was likely responsible for these different responses.

Predicting Possible Future Trajectories for the Southern Nevada Complex

It is very difficult to predict what the future post-fire plant community makeup will look like on the SNC. This is due to the high variability of dominance within the burned areas and the drastic changes from year to year.

Grass-Fire Cycle. One of the main reasons for seeding on the SNC was to prevent a grass-fire cycle from occurring, where increases in highly flammable fuel loads occur due to post-disturbance annual grass dominance, initiating a positive feedback loop in which fire maintains an annual grass-dominated system that continues to burn frequently. Indeed, some of the areas burned by the SNC had burned the previous year (e.g. the Riggs Fire).

For a grass-fire cycle to be occurring in the region, the assumptions of the grass-fire cycle (D'Antonio and Vitousek 1992; Rossiter et al. 2003) must be met: 1) the non-native grasses must change fuel loads post-fire by either increasing fuel loads or increasing flammability; 2) altered fuel characteristics must lead to increased fire frequency; 3) changes in fire frequency should decrease the cover of native plants; and 4) there is an increase in non-native grass dominance post-fire. This study did not specifically test these assumptions. But based on data from the AA plots, annual grass dominance has declined since the first growing season post-fire. While many of the areas adjacent to the SNC burned in 2006 (e.g. Cedar, Clover, Texas, and Sasquatch fires (Figure 7-8), no portion of the SNC has reburned. The high abundance of annual grasses that fueled the SNC fires was initiated by an abnormally wet year. It may be that these types of precipitation patterns and subsequent increases in annual grasses in the next few growing seasons do not happen often enough in the Great Basin-Mojave Transition Zone to fuel a grass-fire cycle. We are still early in the post-fire cycle and how the SNC responds to climatic conditions and fire in the future remains to be seen.

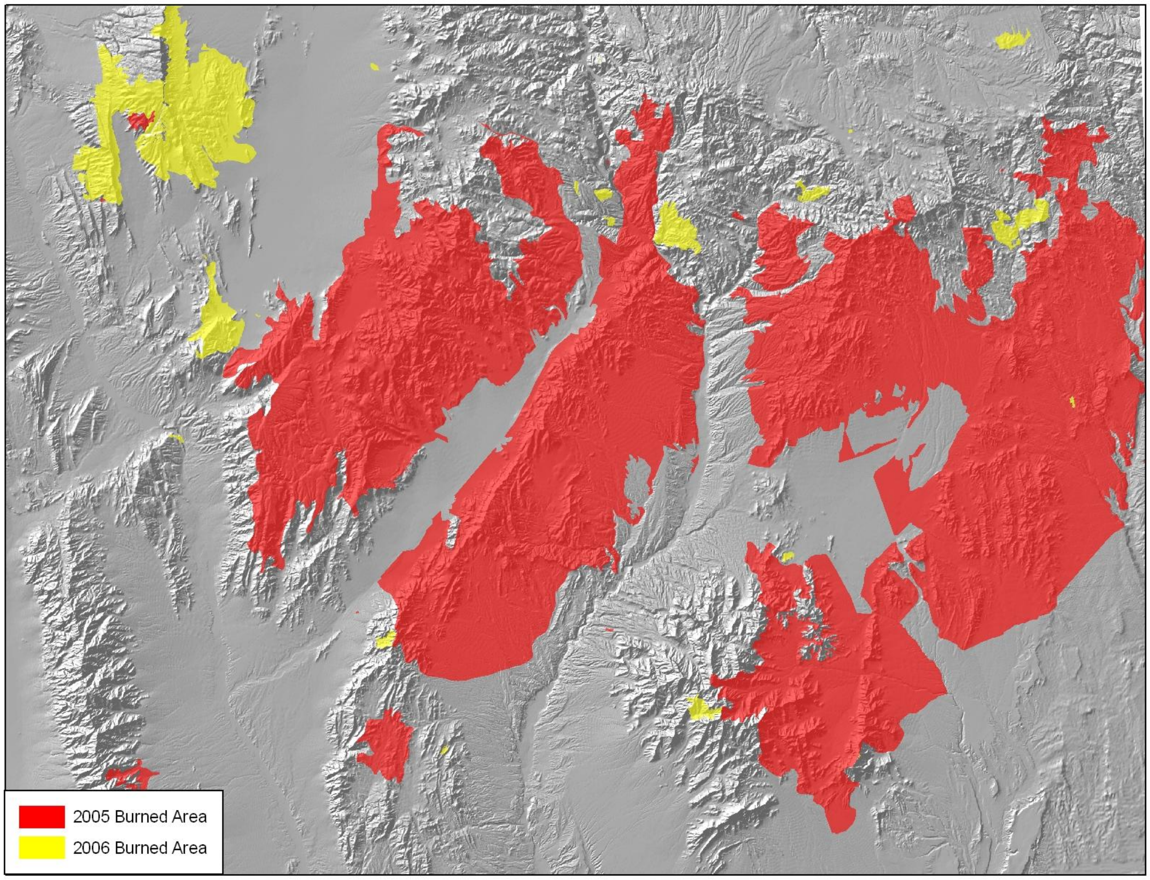

Figure 7-8. Distribution of fires in area of the Southern Nevada Complex in 2005 and 2006.

Resiliency at Higher Elevations. Many of the perennial species establishing in both of the PJ seed mixes are resprouters and are fairly resilient to fire. These species may be increasing in abundance from pre-fire conditions since the non-resilient species cannot return as quickly. It is possible that the higher elevation areas that burned may be transitioning to plant communities that are more resilient to fire. Fire-adapted species such as *Quercus turbinella*, *Eriodictyon angustifolium*, *Rhus trilobata*, and *Amelanchier utahensis* are likely increasing in abundance. Sites that were formerly pinyon-juniper woodlands may become interior chaparral sites. Interior chaparral-dominated sites should be more resilient to fire.

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Chapter 8: Soil Erosion Risks Following the 2005 Southern Nevada Fire Complex

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INTRODUCTION

Management treatments to reduce potential soil erosion are often the main emphasis of post-fire rehabilitation plans, and considerable time, money, and labor are spent on treatments such as vegetation seeding and physical erosion barriers. Unfortunately, the efficacy and cost/benefits of erosion mitigation treatments are rarely evaluated (Robichaud et al. 2000, General Accounting Office 2003), largely because directly measuring soil erosion is labor intensive and expensive. As an alternative, Herrick et al. (2005b) recommend monitoring several indicators of soil erosion potential, specifically cover of perennial and annual plants, rocks, litter, and the surface basal gap area. Basal gap is the area between the rooted stems of perennial plants and primarily influences the potential for erosion caused by flowing water. Plant canopy cover is important for reducing erosion caused by wind. Other soil surface cover, such as rocks and litter, also help reduce the susceptibility of soils to wind and water erosion. Part of the Bureau of Land Management, Ely Field Office rehabilitation monitoring plan specifically calls for measuring properties that may influence erosion, especially soil surface cover.

One of the primary land management concerns in the Mojave Desert is the potential for increased dominance by non-native annual grasses following wildfires. These concerns are largely focused on competition of these non-native grasses with native plants, their effects on fire regimes, and their cumulative effects on wildlife habitat (Brooks and Pyke 2001, Brooks and Esque 2002). The post-fire dominance of non-native annual grasses and their relationship to soil erosion remains largely unevaluated in the Mojave Desert. Some have suggested that erosion within annual dominated communities may increase because of decreases in soil biological crust cover, changes in soil texture, smoother soil surface roughness, and lower water infiltration rates in the adjacent Great Basin ecosystem (Ponzetti et al. 2007, Boxell and Drohan 2009). On the other hand, others have suggested that the quick recovery of annual plants following a fire may provide cover that will reduce erosion in the short-term until perennial plant cover re-establishes (Klemmedson and Smith 1964, Rickard and Warren 1981). In this chapter we evaluate the indirect evidence for effects on soil erosion potential following the 2005 Southern Nevada Fire Complex.

METHODS

Site Description and Study Design

The study area was located within and adjacent to portions of the 2005 Southern Nevada Fire Complex primarily within Lincoln County, Nevada, in the eastern Mojave Desert (Webb et al. 2009). The vegetation types evaluated were pinyon-juniper woodlands within wilderness and non-wilderness areas, and blackbrush shrublands. General characteristics of the 2005 Southern Nevada Fire Complex, a more detailed description of the study areas, and

explanation of the monitoring design and the BB macro-plot data that were used in the current chapter are discussed at length in Chapters 1 and 4 of this report. Soil erosion risk indicators that were quantified at each macro-plot included the coverage of basal gaps, and the canopy cover of living perennial vegetation, annual vegetation, and litter. Sampling was done during the spring and summer of postfire years 1, 2, and 3 (2006-2008)

Sampling Methods

Basal gap was quantified using a line-intercept technique (Herrick et al. 2005a) along the two 30-m edges of each BB macro-plot (Chapter 4). The total length of individual gaps exceeding 20 cm between rooted perennial plants (either dead or living) was summed along the two transects then divided by the total transect length and multiplied by 100 to arrive at a percentage. These methods were used to quantify basal gaps in unburned and burned areas during post-fire year 1 (2006) and in burned areas during postfire years 2 and 3. Postfire data represent repeated measures of the same plots. Unburned data were collected concurrently with the post-fire year 1 data, but from plots located outside of the burn perimeter that were matched with burned plots based on proximity, vegetation type, soil type, topography, and elevation.

Cover was measured using a point-intercept method in the unburned and post-fire year 1 samples. Methods were changed to ocular estimates of cover during post-fire years 2 and 3 (see Chapter 4 for more details). It is possible that different values could be derived using these two methods, so we derived the final values used in our analyses as follows. Actual point cover data were used for the unburned and post-fire year 1 levels. Derived point cover data were used to characterize post-fire years 2 and 3. These values were derived from relationships between point cover and density data collected during post-fire year 1 and in a subset of plots collected during post-fire year 2. These positive relationships were found to be statistically significant, and because density was collected in the same manner over all post-fire years we were able to produce a derived set of point cover data for the final two years (see Chapter 6 for more details). Cover data in the burned area were repeated measures of the same plots, whereas unburned data were collected from adjacent matched plots outside of the fire perimeter.

Data Analysis

Data were statistically analyzed using Bayesian hierarchical models (McCarthy 2007). The explanatory variables included 3 levels of vegetation type/seed mix (blackbrush, non-wilderness pinyon-juniper, and wilderness pinyon-juniper), 2 levels of seeding (seeded and control), time (years 1–3 for basal gap and years 2–3 for the other cover categories because of a change in methodology from a point-intercept technique in year 1 to an ocular estimation technique in years 2–3), percent slope, plus interactions among the explanatory variables. The response variables were percent basal gap cover, live perennial vegetation cover (sum of

all living perennial grasses, forbs, shrubs, and trees), annual vegetation cover (sum of all annual grasses and forbs), and litter (sum of litter and duff). Cover values were logit transformed prior to analyses using the function $\log_e$((percent cover/100)/(1-(percent cover/100)). Since the sampling design included subsampling (multiple macro-plots within a larger seeding treatment plot), spatial clustering (seeding treatment plots clustered throughout the landscape), and repeated sampling over years, the models also included random effects which consisted of a cluster effect (a unique index for each spatial cluster) and a plot nested within cluster effect.

Model parameters were calculated by generating their posterior probability distribution by creating a Markov chain/Monte Carlo sample using the RJAGS package (Plummer 2009) within the R statistical software system (R Development Core Team 2008). Informative priors were not included in the analyses because comparable data were not readily available on immediate post-fire responses of blackbrush and pinyon-juniper vegetation communities in this particular region. The posterior distributions from this study can be used as priors for future analyses, especially for monitoring data Ely BLM are collecting for subsequent wildfires. Parameters with 95% credible intervals that did not include 0 and ecologically meaningful effect sizes were judged to be the most influential. In addition to the post-fire seeding experiment, we also analyzed data from unburned plots established and monitored by USGS. The same sampling methods were used for these plots, but data were only collected for 1 year.

RESULTS AND DISCUSSION

Basal gaps were not affected immediately following fire, although there was some indication of a latent decline by postfire year 3 in the wilderness and non-wilderness pinyon-juniper plots (Figure 8-1). This latent decline was likely due to increased stem density of perennial plants and/or increased diameter of basal areas of perennials (e.g. perennial grasses and basal sprouting by perennial shrubs). Even if this latent decline in basal gaps is considered statistically significant, it only amounts to about a 1% decrease from an already very high preburn basal gap cover (about 97% in blackbrush and 98% in pinyon-juniper plots). There are very little data quantifying the relationships between changes in basal gap and actual soil erosion (Herrick et al. 2005b), but even without this information one must question the ecological significance of a 1% change in basal gap cover. It seems safe to assume that in the absence of other information the changes in basal gap observed following the 2005 Southern Nevada Complex probably have had a negligible effect on soil erosion potential.

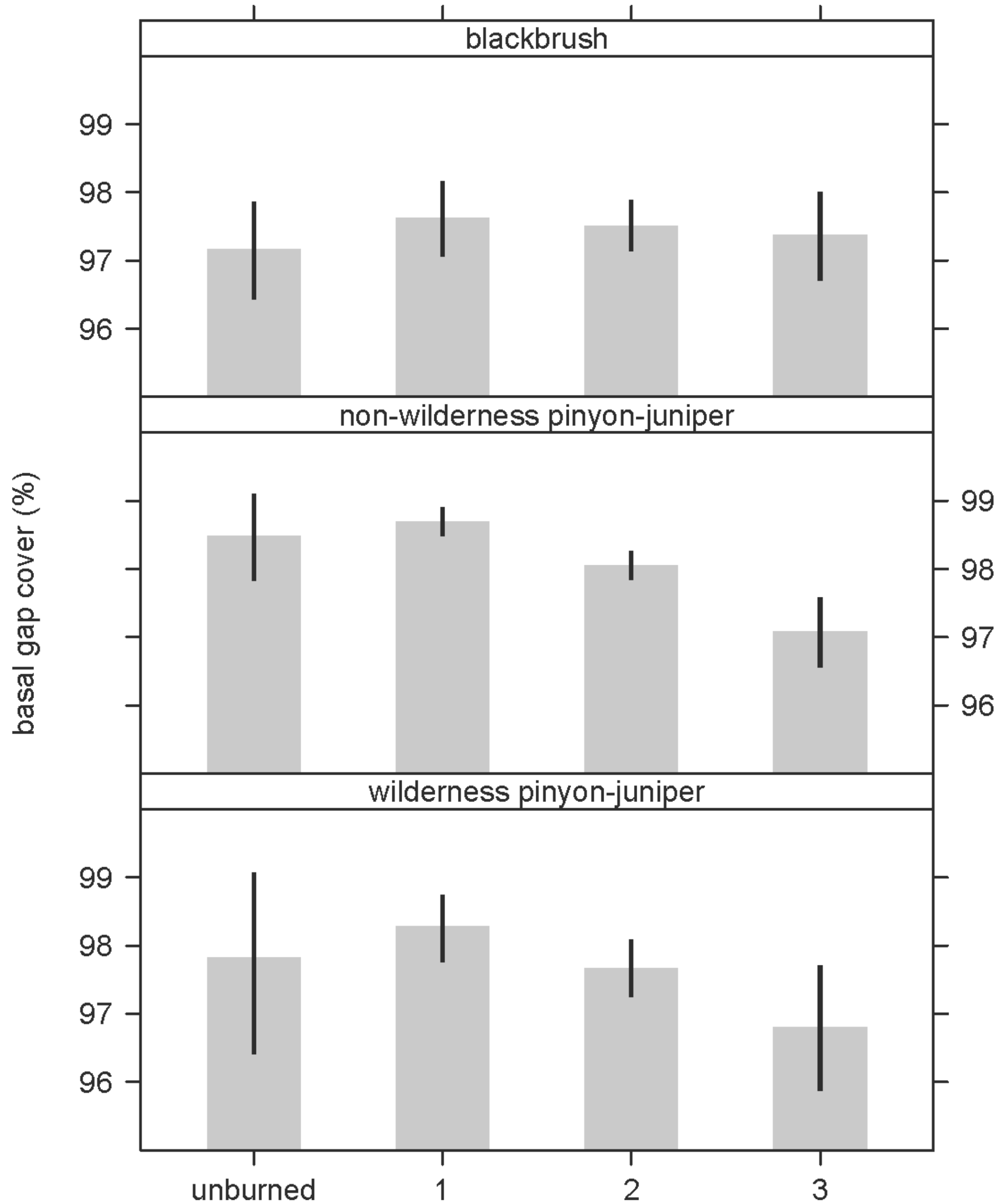

Figure 8-1. Basal gap between the rooting axes of perennial plants in unburned areas and burned areas during posfire years 1-3. The error bar represents the 95% credible interval (Bayesian analog for confidence interval).

Annual plant cover was similar in burned and unburned areas during post-fire year 1 in the blackbrush plots, but was significantly lower in burned areas in the pinyon-juniper plots (Figure 8-2). Annual plant populations in the Mojave Desert often decline the first year following fire, but often rebound in terms of cover to levels at or above pre-fire conditions by the second year (M. Brooks in prep). However, without comparable unburned plots during postfire years 2 and 3, it is impossible to determine if annual plant cover remained lower in burned than unburned plots during those years. Productivity by annual plants can vary widely based on annual rainfall amounts in the Mojave Desert (Beatley 1974), and any attempt to evaluate trends following fire must incorporate comparisons with comparable unburned areas during each of the year of interest. Nevertheless, these current and past results suggest that concerns about total annual plant cover being reduced following fire may be warranted during the first post-fire year at least in the higher elevation pinyon-juniper zones, but probably not beyond the first post-fire year.

Perennial plant cover was greatly reduced during the first post-fire year in both blackbrush and pinyon-juniper vegetation types (Figure 8-3). Cover remained low through post-fire year 3 in blackbrush, but showed some small signs of recovering in pinyon-juniper (Figure 8-3). Within blackbrush ecosystems, recovery of perennial shrub cover to pre-fire levels can take well over a decade or two, although the species composition remains altered for longer time periods (Callison et al. 1985, Brooks and Matchett 2003). Perennial cover at higher elevations in pinyon-juniper vegetation types can recover much more rapidly (Brooks et al. 2007). Reduced cover has major implications for wind erosion (Herrik et al. 2005b). Although quick recovery of annual plant cover may help mitigate some of the effects of perennial cover loss, annuals cannot replace the coarse physical structure and windbreaks that only perennial plants provide.

Cover of litter and duff was significantly reduced due to combustion during the fires (Figure 8-4). The greatest declines occurred in pinyon-juniper where downed woody material covered over half of the soil surface in unburned areas. The only recovery was observed by post-fire year 3 in blackbrush where litter from non-native annual grasses showed increases (Figure 8-4 and M. Brooks pers. obs.).

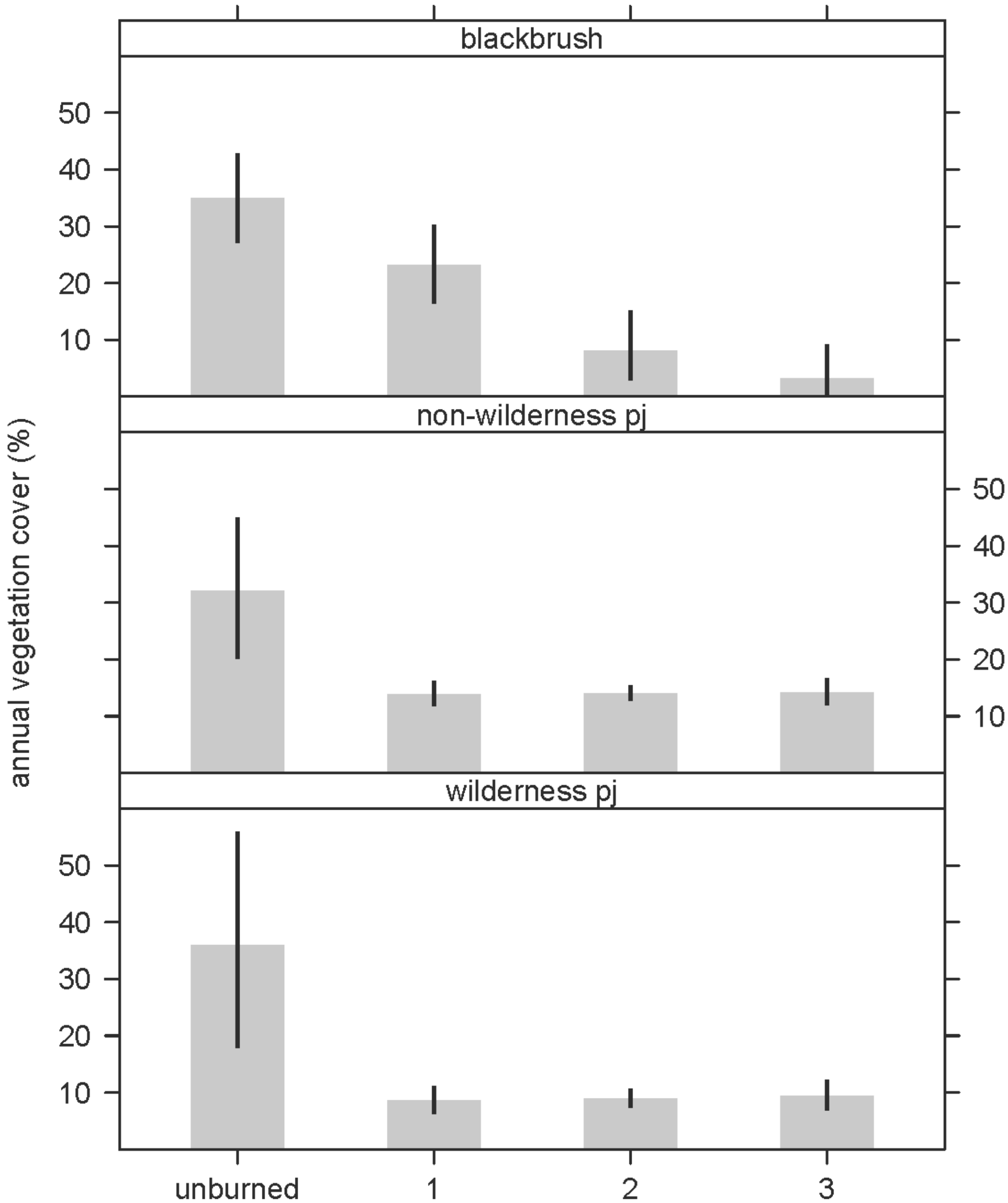

Figure 8-2. Annual plant cover in burned and unburned areas during postfire year 1, and in burned areas between post-fire years 2 and 3. The error bar represents the 95% credible interval (Bayesian analog for confidence interval).

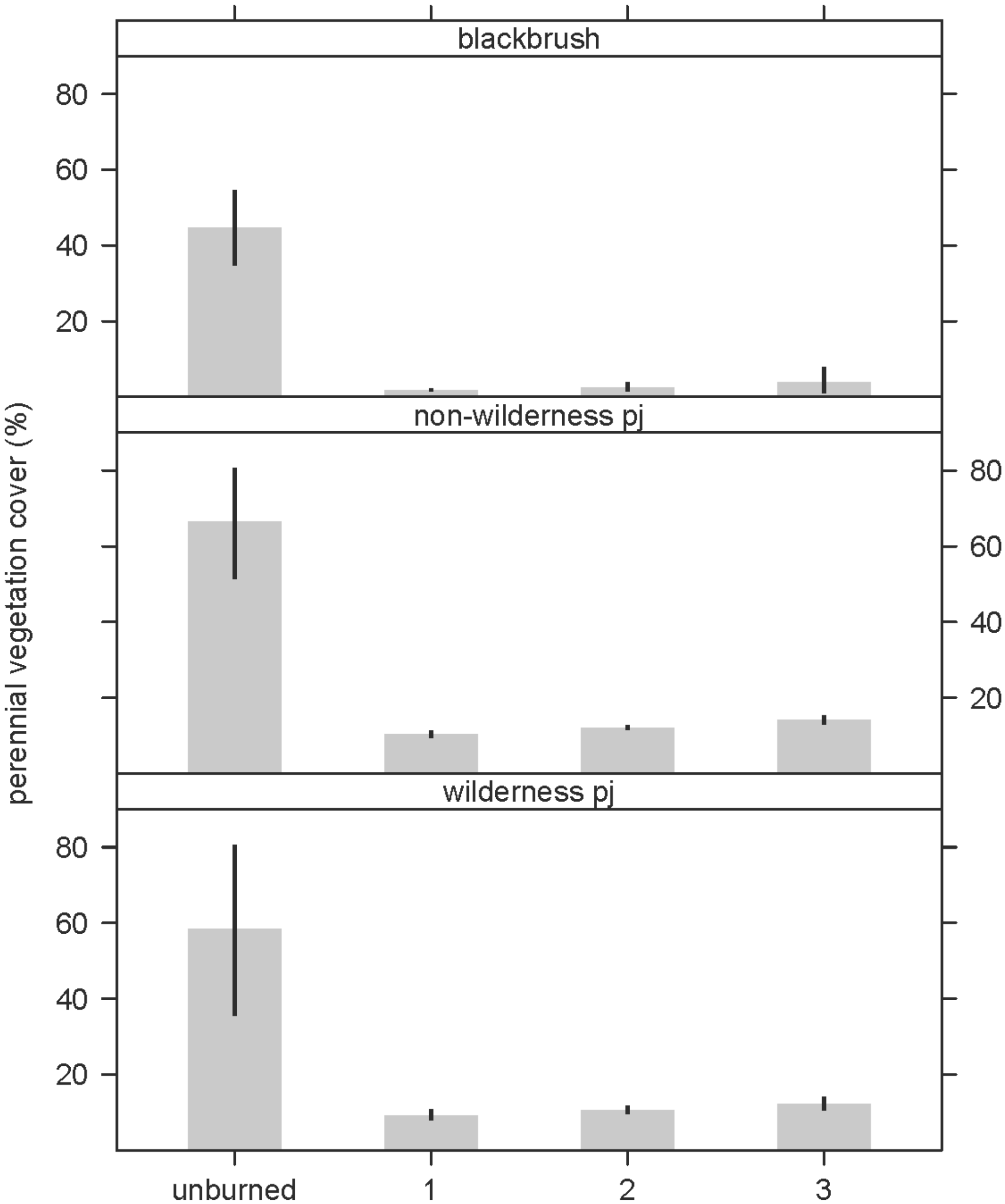

Figure 8-3. Perennial plant cover in burned and unburned areas during postfire year 1, and in burned areas between post-fire years 2 and 3. The error bar represents the 95% credible interval (Bayesian analog for confidence interval).

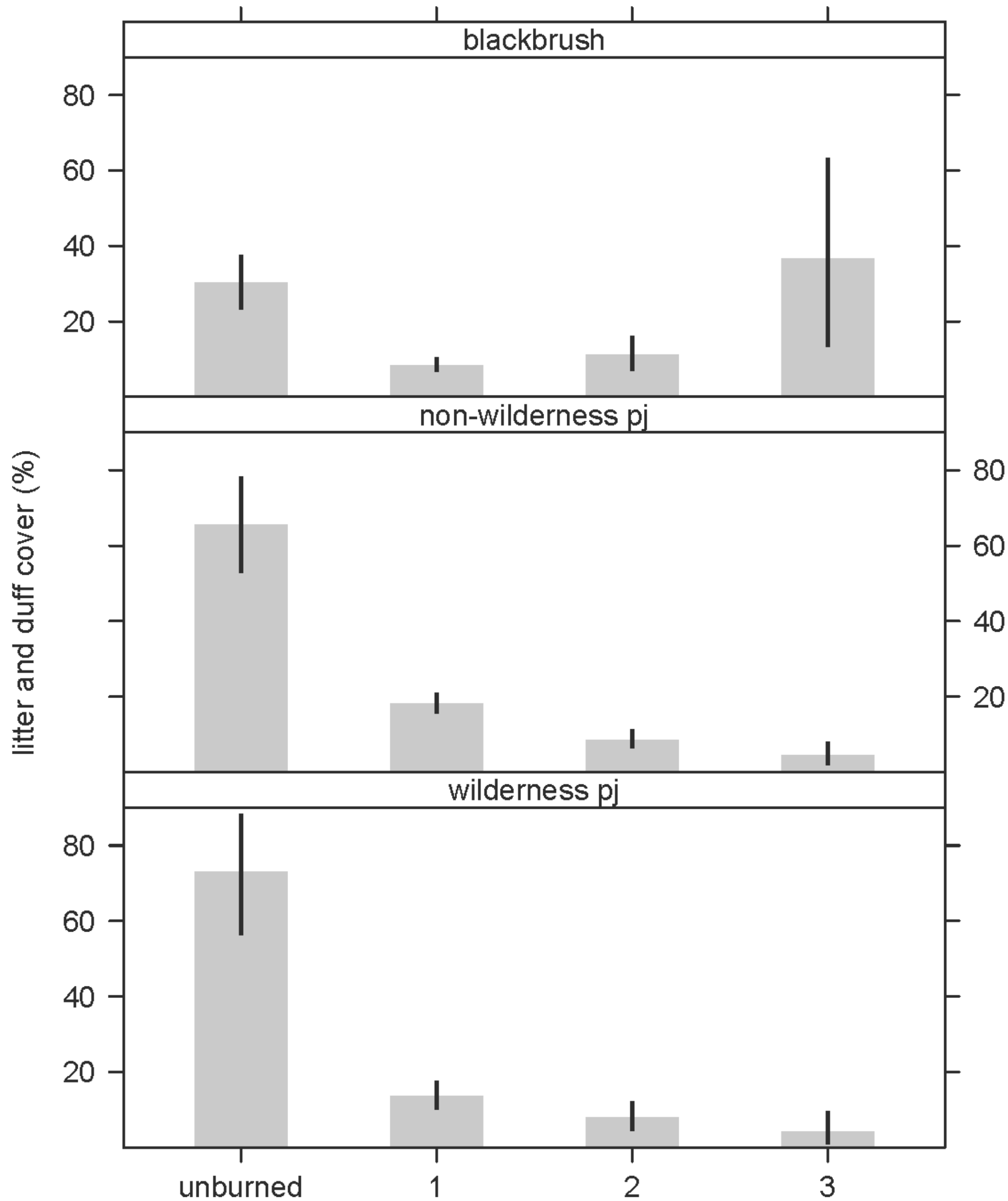

Figure 8-4. Litter and duff cover in burned and unburned areas during postfire year 1, and in burned areas between post-fire years 2 and 3. The error bar represents the 95% credible interval (Bayesian analog for confidence interval).

MANAGEMENT IMPLICATIONS

There were no notable effects of fire on basal gaps between perennial plants, which is primarily relevant to water erosion. Basal gaps in unburned areas already comprise about 98% of the total ground surface, so there is little room for further reductions due to fire anyways.

Annual plant cover is also relevant to water erosion, and was significantly reduced during postfire year 1 in both wilderness and non-wilderness pinyon-juniper vegetation, but not in mesic blackbrush. If water erosion is a concern, then seedings that include annual species may be warranted within burned pinyon-juniper vegetation during the first postfire year. Unfortunately, the longer-term effects of fire on annual plant cover, and the potential value of seedings during subsequent postfire years, cannot be evaluated due to the absence of unburned data beyond the first postfire year.

There were much more notable declines in perennial cover following fire, especially in blackbrush vegetation, which may have affected wind erosion, and in litter and duff cover, especially in pinyon-juniper vegetation, which could have affected both wind and water erosion. Perennial vegetation in particular may take many decades to re-establish following Mojave Desert fires, although recovery of species composition often takes much longer (Brooks and Minnich 2006). Seeding of perennial species may be warranted to restore perennial cover following fire, however as indicated in Chapter 6 establishment of seeded perennials is only likely during years of high rainfall and where non-native annual plant density is low.

Overall seeding effects were negligible during the first three post-fire years (see Chapter 5), so the results presented in the current chapter regarding soil erosion potential should be interpreted as basic postfire effects.

The links between the effects of fire on the vegetation characteristics reported in this chapter and their effects on soil erosion potential are not specifically understood. This information should be generated through future research and monitoring efforts to evaluate the ultimate value of post-fire seeding for the purposes of reducing soil erosion in the Mojave Desert.

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Chapter 9: Remote Sensing Assessments and Applications

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INTRODUCTION

In June and July of 2005, dry lightning storms ignited eleven fires and burned approximately 740,000 acres in the Bureau of Land Management (BLM) Ely and Las Vegas Districts. For fire management reasons, these fires were combined and named the Southern Nevada Complex (SNC). Remote sensing technologies were initially utilized in July 2005 at the request of the Department of Interior (DOI) Burned Area Emergency Response (BAER) team. The team requested standard Burned Area Reflectance Classification (BARC) products, which are based upon Landsat Thematic Mapper (TM) satellite imagery. The standard BARC product suite has been defined by the U.S. Forest Service Remote Sensing Applications Center (USFS RSAC) and U.S. Geological Survey Center for Earth Resources Observation and Science (USGS EROS) (USFS RSAC, 2009). These products were generated immediately upon containment of the SNC fire and provided to the BAER team. The BAER team used these products to generate the final soil burn severity map used for many subsequent burn area analyses. After completion of the BAER analysis, the BLM Ely Field Office requested a more in-depth assessment of remote sensing capabilities (with emphasis on the Landsat Thematic Mapper 30 meter resolution sensor) related to burn severity assessments, the selection of post-fire seeding locations, assessments of seeding effectiveness, assessments of general burned area vegetation recovery, mapping the occurrence of invasive annual and perennial plants, and other applications potentially relevant to the management of lands impacted by the SNC. The BLM Ely District and USGS EROS entered into an agreement to address these issues for the SNC in FY2006. Subsequent agreement modifications have extended these analyses to other fires and other related tasks that are still in progress. This report summarizes results only associated with the SNC.

STUDY AREA

To fully assess the capability of moderate resolution Landsat satellite imagery to satisfy BLM burned area mapping and monitoring objectives for the SNC, a time series of Landsat 5 images were acquired for the SNC study area (Figure 9-1). The SNC included burned areas in the BLM Las Vegas District as well as the primary burn area to the north located in the BLM Ely District. A large portion of Nevada’s Lincoln and Clark counties were within the study area. Analysis was limited to areas within the final SNC fire perimeters. In addition to Landsat satellite imagery, other Geographic Information System (GIS) data layers were developed or acquired to assist in meeting the overall remote sensing project objectives. The data layers acquired or derived for the project are described in detail in the Data section of this report.

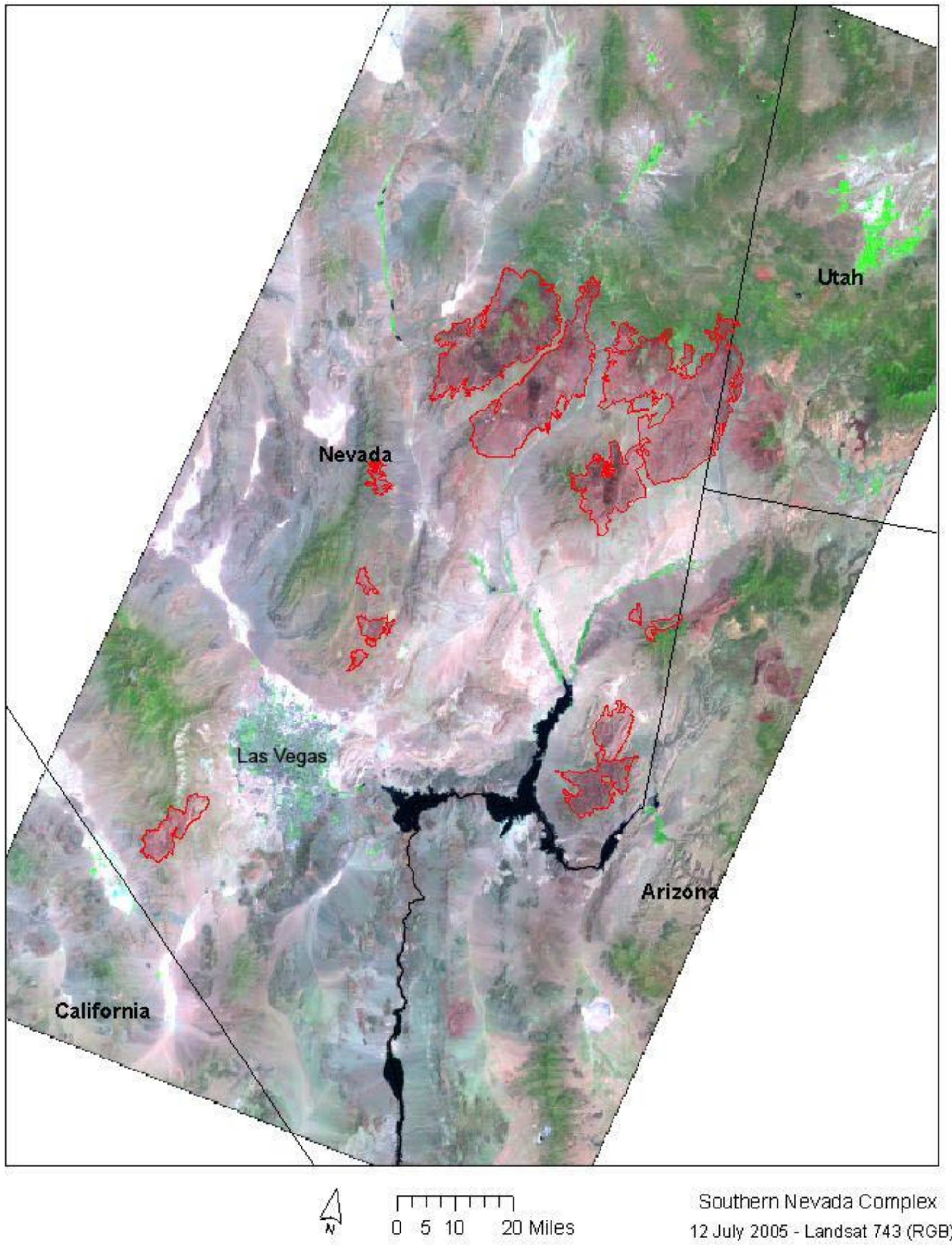

Figure 9-1. Project study area located in southeast Nevada. Remote sensing analyses were limited to a time series of Landsat satellite images acquired for path 39/ rows 34 and 35. This provided coverage for all Southern Nevada Complex (SNC) fire perimeters as depicted in red. The image above was acquired on 12 July 2005 immediately after containment of fires. Burned areas appear in shades of red within the red SNC fire perimeter polygons.

DATA

Numerous ancillary geospatial data layers were acquired to support the remote sensing effort for the SNC. Vector and raster GIS data layers related to sampling point locations, seeded and non-seeded polygons, roads, allotment boundaries, and many other themes were provided by the BLM Ely District office. USGS DEM, NAIP, political boundaries, and other GIS data layers were acquired from state and national online data clearinghouses. The primary data layers derived by this project and related directly to the analysis of the burn area included the Landsat time series, NOAA precipitation, and historical fire/burn information.

Landsat Time Series

The original project plan called for the acquisition of Landsat 5 satellite imagery for the SNC study area on approximately a monthly basis starting in May 2005 and continuing through September 2008. Due to clouds and/or snow cover and Landsat 5 satellite technical issues, image acquisition proved possible for 31 of the 41 targeted monthly periods. This resulted in the acquisition of over 70 scenes primarily for Landsat path 39 and rows 34 and 35 (Table 9-1).

Of the missing images in the Landsat monthly time series, only 5 of the dates were outside of the November to January time frame where snow cover or reduced vegetation vigor reduced anticipated information content. Of more concern were the six missing dates near or at potential phenological peaks of green vegetation. For example, the consecutive months of March and April of 2006, and February of 2007 were considered important dates usually coinciding with the peak of greenness for annual grass. A four month gap in coverage for the period October 2007 through January 2008 was directly due to the temporary loss of the Landsat 5 satellite due to a technical sensor malfunction. In general, all imagery acquired was high quality and consistently cloud and snow free for the study area.

There is potential to fill Landsat data gaps in the SNC time series by compositing multiple Landsat 5 scenes possessing varying levels of cloud/shadow impacts with scenes derived from Landsat 7. Landsat 7 images were originally avoided as they possess a data gap anomaly due to the satellite scan line corrector (SLC) problem which has affected Landsat 7 imagery since May 31, 2003. Landsat 7 images are difficult to use when needing full and actual (not estimated values) image content. However, cost of imagery was a consideration at the time of the acquisition of Landsat scenes for the SNC. Purchasing several scenes to generate one cloud free composite scene was not a viable solution. The U.S. Geological Survey (USGS) recently began offering no-cost Landsat data. In the future, supplementing the SNC time series by compositing multiple marginal scenes within critical time periods is an option.

Table 9-1. Landsat 5 time series coverage dates for path 39/rows 34 and 35.

Target Period	Date of Coverage
May 2005	5/25/2005
June 2005	6/26/2005
July 2005	7/12/2005
August 2005	8/29/2005
September 2005	9/14/2009
October 2005	No Coverage
November 2005	11/17/2005
December 2005	No Coverage
January 2006	1/20/2006
February 2006	2/21/2006
March 2006	No Coverage
April 2006	No Coverage
May 2006	5/12/2006
June 2006	6/29/2006
July 2006	7/15/2006
August 2006	8/16/2006
September 2006	9/17/2006
October 2006	10/19/2006
November 2006	11/20/2006
December 2006	12/06/2006
January 2007	1/23/2007
February 2007	No Coverage
March 2007	3/12/2007
April 2007	4/29/2007
May 2007	5/15/2007
June 2007	6/16/2007
July 2007	7/18/2007
August 2007	8/19&3/2007
September 2007	9/4/2007
October 2007	No Coverage
November 2007	No Coverage
December 2007	No Coverage
January 2008	No Coverage
February 2008	2/27/2008
March 2008	3/14/2008
April 2008	4/15/2008
May 2008	5/17/2008
June 2008	6/18/2008
July 2008	No Coverage
August 2008	8/21/2008
September 2008	9/6/2008

Two Landsat path/row footprints provided full coverage of the SNC. The primary path/row was 39/34, while a secondary path/row of 39/35 provided coverage for the southern portion of the study area and the BLM Las Vegas District. A total of 37 Landsat 5 scenes were acquired for 39/34 while 34 scenes were acquired for 39/35. Additionally, several Landsat 5 and 7 scenes were acquired for primary and adjacent path/rows for general

reference, but not used in subsequent analyses. In all but one case (August 2007), imagery was acquired on the same date (same satellite pass or swath) for both the 39/34 and 39/35 path/row pairs. These image pairs were mosaicked and subset to the SNC to provide single-image coverage of the study area. For August 2007, due to cloud issues, images from 19 August 2007 and 03 August 2007 were mosaicked to provide study area coverage.

Only Landsat 5 Thematic Mapper (TM) scenes and spectral bands 1-5 and 7 were used in SNC mapping and monitoring analyses. All Landsat 5 TM scenes were terrain corrected and converted to at-satellite reflectance for the six reflective bands based upon the Multi-Resolution Land Characteristics (MRLC) image protocol (Homer et al., 2004). The conversion algorithm is physically based, automated, and does not introduce significant errors to the data (Homer et al., 2004 and Huang et al., 2002).

NOAA Precipitation

National Oceanic and Atmospheric Administration (NOAA) National Weather Service (NWS) Precipitation Analysis data were acquired for the conterminous USA with data organization and formatting assistance from the USGS Center for EROS. These data provide gridded precipitation estimates for the study area on a daily and monthly summary basis. Precipitation data extends from January 2005 through October 2008. The spatial resolution of these data is four kilometers. The data were derived by the NWS using a multi-sensor approach, involving the comparison of WSR-88D NEXRAD estimates to ground rainfall gauge reports (NOAA NWS, 2009). Applications of these data to the SNC and related analyses are reported in Chapter 6 of this report. At this time these data are experimental. As the strengths and weaknesses of this data set become better known, it will likely be used more in the future.

Historical Burn Data

The SNC study area experienced multiple wildfires prior to 2005. Historical fire perimeters were collected from local, state and federal agency sources and originally derived using a number of methods: GPS, manual interpretation of day time or night time (thermal infrared) images, sketch mapping from aircraft, and other methods. When possible, perimeter and soil burn severity estimates from satellite imagery were collected for historical fires. These satellite derived products were primarily collected from two USGS/USFS sources. The joint USGS/USFS Monitoring Trends in Burn Severity (MTBS) project provided historical burn information for the period of 1984 through 2006 (USGS/USFS 2009). The joint BAER emergency response burn mapping support service provided data for fires mapped from 2003 to 2008. Both sources provided GIS compatible historical burn information consisting of pre-fire and post-fire satellite imagery, classified soil burn severity, raw burn ratio layers and fire perimeter vector data. Details concerning the use of historical burn data in the SNC project are provided in the following report RESULTS section.

METHODS

We developed methods to use Landsat 5 multitemporal satellite images with results from rigorous ground sampling to assess and map vegetation greenness, vegetation recovery, seeding effectiveness, soil burn severity and vegetation mortality, and annual/perennial plant dominance for the SNC. Specific methodological descriptions of tools, techniques used, and products generated are detailed in following sections.

Landsat Time Series Analysis

The multitemporal satellite image database developed for the SNC allows the analysis of satellite image data or derived indices at varying time intervals from immediate pre-fire 2005 (May 2005) to over 3 years post-fire 2008 (September 2008). Satellite image indices or individual spectral band values may be extracted at field macroplot point coordinates, by paired-plot polygons, or other user-defined geographic areas. The extraction of image data values from 30 meter Landsat data were generally accomplished using a bilinear approach to image sampling. Briefly, this involves identifying a two by two pixel area, nearest the target macroplot point or coordinate, and to then calculate a distance-weighted output value. Extracted image values, along with field data parameters, were organized in a spreadsheet or database with records based upon the 1,173 macroplots. Further manipulation of the time series data for macroplots was accomplished in the database and spreadsheet environment. For example, ratio or index differences were calculated in a spreadsheet versus being derived through digital image processing methods. When image or map products were required, processing was accomplished using image processing software.

Burn Severity Indices

The Normalized Burn Ratio (NBR), differenced NBR (dNBR) and the Relative differenced NBR (RdNBR) were all used to map soil burn severity and vegetation mortality for the SNC. In very general terms, the dNBR was used to assess the absolute loss of biomass and soil burn severity which have been correlated with soil impacts due to fire. The RdNBR was used for assessments when the objective was to assess relative amounts of vegetation mortality. These indices are described in the following section.

NBR and dNBR. The Normalized Burn Ratio (NBR) is computed using Landsat ETM/TM near infrared (NIR) and short wave infrared (SWIR) spectral bands (4 and 7). The NBR is calculated as follows:

$$NBR = (NIR - SWIR)/(NIR + SWIR)$$

For burn severity mapping purposes the NBR is generally calculated for both a pre-fire and post-fire image and then used to derive a differenced NBR (dNBR) as follows:

$$dNBR = NBR_{prefire} - NBR_{postfire}$$

After being developed by Key and Benson (2005a), NBR and dNBR have been widely used to map burned areas across the USA and internationally.

RdNBR. Miller and Thode (2007) proposed the relative differenced NBR (RdNBR) to remove the biasing of the pre-fire vegetation by dividing dNBR by the square-root of the pre-fire NBR as follows:

$$RdNBR = (NBR_{prefire} - NBR_{postfire}) / \sqrt{(|NBR_{prefire}| / 1000)}$$

By convention, NBR is normally scaled by 1000 to transform the data to integer format; therefore the pre-fire NBR must be divided by 1000 in the RdNBR formula (Miller et al., 2009).

Vegetation Indices

NDVI and dNDVI. The Normalized Difference Vegetation Index (NDVI) is an index widely used to identify vegetated areas and to estimate their condition since the early 1970's (Rouse et al., 1973). It is also a standardized method of comparing vegetation greenness between satellite images. The NDVI is calculated as follows:

$$NDVI = (NIR - RED)/(NIR + RED)$$

NDVI values can range from -1.0 to 1.0. Increases in the value are associated with increases in levels of healthy vegetation cover. NDVI values near zero and decreasing negative values indicate non-vegetated features such as barren surfaces (rock and soil) and water, snow, ice, and clouds. Rock and bare soil values tend to range close to zero, while water bodies tend to have negative values. Clouds and snow will cause values near zero leading to the appearance that vegetation is less green. In a rangeland environment, NDVI is influenced by a mixture of cover types such as green healthy vegetation, dead vegetation, and soil. The combination of these cover types will determine the NDVI value. A pixel dominated by high density green grass and/or green leaves may have a NDVI value approaching 0.8 and higher, whereas a pixel dominated by dead grass or dry soil may have values in the 0.12 to 0.3 range.

When mapping burn severity with imagery lacking a SWIR band (a band used in the dNBR calculation) NDVI has been shown to be a suitable substitute for NBR and dNBR (Hudak, 2007). For burn severity mapping or change detection purposes, the differenced NDVI (dNDVI) is calculated using two images acquired at different time intervals (i.e. pre-fire and post-fire) as shown in this example for mapping burn severity:

$$dNDVI = NDVI_{prefire} - NDVI_{postfire}$$

Field Data for Remote Sensing

Three plot types (data collection procedures) were used by field crews including “AA” or additional aerial seeding plots, “BB” or brushbelt plots, and “RS” or remote sensing plots with only ocular cover estimates (Figures 9-2 and 9-3). All field macroplots, regardless of plot type were generally included in remote sensing analyses. Not all plots were visited every sampling period (2006, 2007 & 2008). The RS plots were added specifically to provide additional burn area vegetation cover information where there were no paired-plots and where there was sparse field data in non-seeded areas. The need for a good distribution of field macroplots across the full SNC and for each sampling period required the use of all plot types, although it was recognized that the rigor of procedures, such as those for canopy cover estimation (transect versus ocular), may vary. A full description of the field sampling design and procedures are provided in Chapter 4 of this report.

Remote sensing analyses used a specific subset of all the variables collected by field crews. These variables included:

- TVGC: Estimate of the total live vascular plant cover for the macroplot. Includes perennials, annuals, and biennials.

- IAGC: Estimate of the total live non-native invasive annual grass cover for the macroplot. Species included are: *Bromus madritensis, Bromus tectorum, Bromus trinii, Bromus arvensis, Schismus arabicus* and *Schismus barbatus*. No other non-native invasive annual grasses have been noted within the study sites.

- PRNC: Estimate of the total live perennial plant cover for the macroplot.

- SHRC: Estimate of the total live shrub and subshrub cover.

- BRRC: Estimate of the total live cover of Bromus madritensis (red brome).

- BRTC: Estimate of the total live cover of Bromus tectorum (cheatgrass).

- EROC: Estimate of the total live cover of Erodium cicutarium (storksbill).

- IAGD: The average density (number of plants per square meter) of invasive, non-native annual grasses within a macroplot. This includes the following species: *Bromus madritensis, Bromus tectorum, Bromus trinii, Bromus arvensis, Schismus arabicus, and Schismus barbatus*. No other non-native invasive annual grasses have been noted within the study sites.

- PRND: The average density (number of plants per square meter) of all living perennial plants within a macroplot, including grasses, forbs, shrubs, subshrubs, and trees.

Statistics

Most assessments of remote sensing data including derived indices were conducted using a simple linear regression approach. For example, when evaluating a derived index (i.e., NDVI, dNBR), the index is commonly the predictor variable (x axis) and the field data value is the dependent variable (y axis). In reporting results from linear regression analyses, an r^2 value is included providing a "goodness-of-fit" measure for the result. The value r^2 is a fraction between 0.0 and 1.0, and has no units. An r^2 value of 0.0 means that there is no linear relationship between X and Y, or the best-fit line is a horizontal line going through the mean of all Y values. When r^2 equals 1.0 all points lie exactly on a straight line with no scatter (GraphPad Software, 2009).

Work is in progress to assess the usefulness of classification tree and regression tree techniques for mapping land cover patterns of interest in the SNC, such as annual and perennial plant distributions. These techniques have been used successfully by USGS national land cover mapping programs (Homer et al. 2004), but generally require the availability of extensive field data to generate maps with reasonable accuracies. For the period of 2006 to 2008, we possess extensive field data (% cover estimates in particular) for the SNC. It is anticipated these techniques will be explored to see if they offer advantages in creating SNC maps of annual/perennial plant canopy cover within the 2006-2008 time frame. Where 2006 Composite Burn Index (CBI) (Key and Benson 2005b) data are also available as a compliment to other field data, additional work is being considered to test the applications of these techniques to the mapping of burn severity and vegetation mortality. Software anticipated to be used for these analyses includes See5, a decision tree program, and Cubist, a regression tree algorithm, both developed by RuleQuest Research (http://www.rulequest.com). In support of national land cover mapping programs (Homer et al. 2004), the ERDAS Imagine Classification and Regression Tree (CART) software module was developed to assist users with the generation of spatial map products by applying classification/decision tree rules developed in See5. This module will also be used in SNC work.

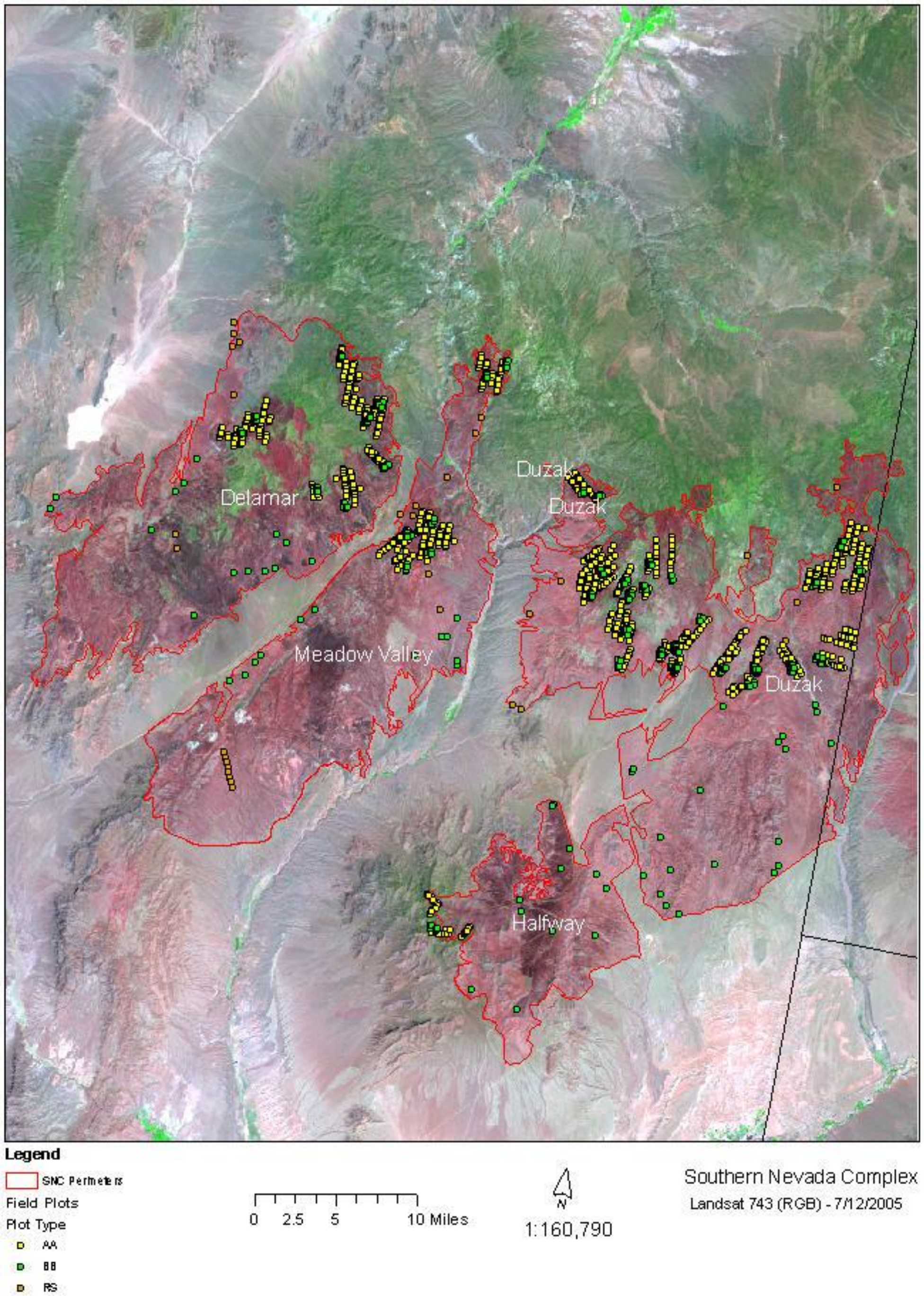

Figure 9-2. Location overview of 1,193 field sampling points. Plot types included AA or additional aerial seeding plot, BB or brush belt plot, and RS or remote sensing plot with only ocular cover estimates.

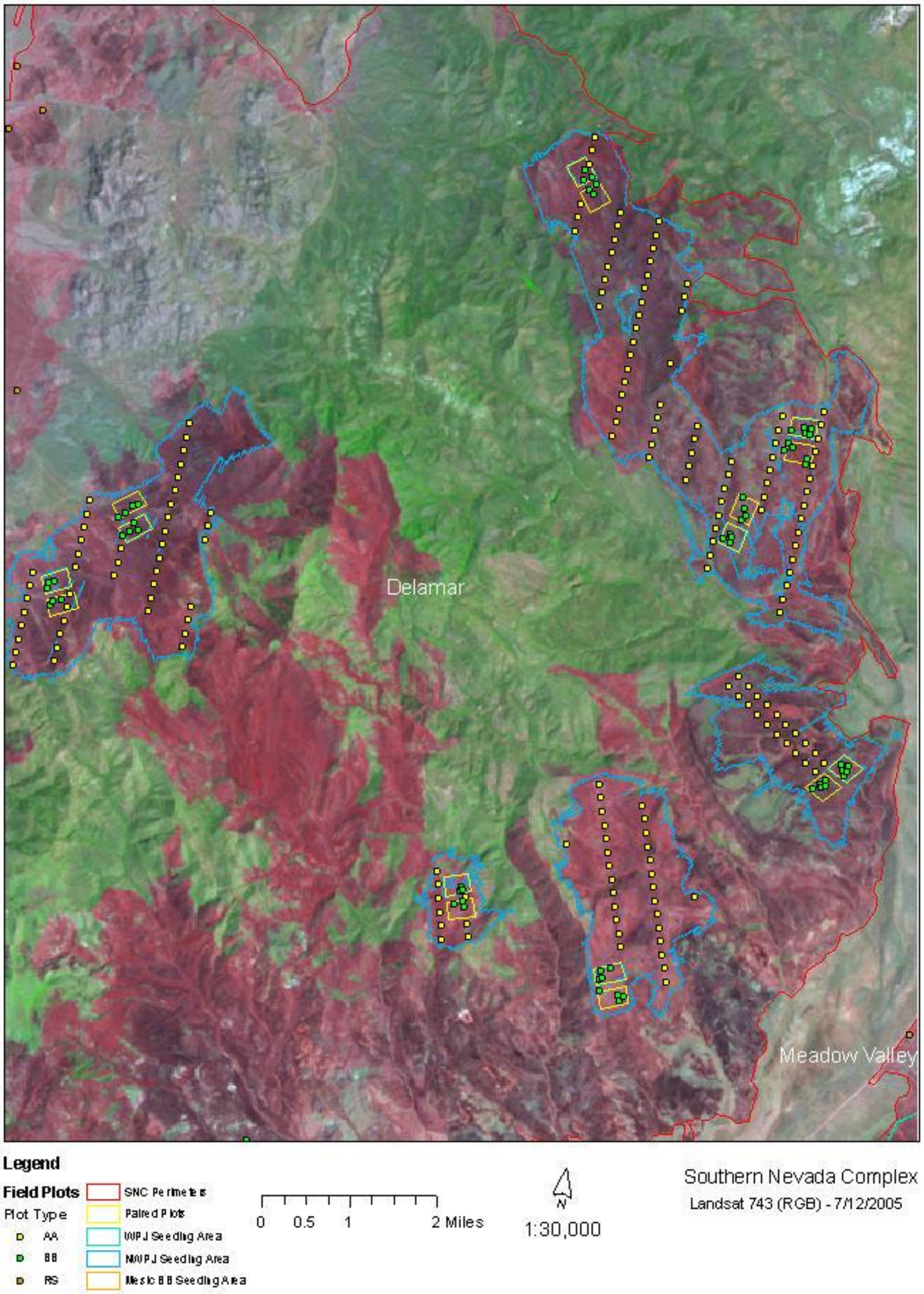

Figure 9-3. Close-up view of field sampling points for portion of the Delamar fire. AA and BB plots are primarily located within the seeding areas. BB plots are focused upon the paired-plots. RS plots are located across the SNC in areas sparsely covered by AA and BB plots.

All statistical work presented in this report was accomplished using a combination of Microsoft (MS) Access 2003 (SP2) and MS Excel 2003 (SP2). Graphs/charts and regression analyses were accomplished using MS Excel. MS Access was used to organize all raw data, conduct standard database queries, and prepare tables for use in Excel.

Image Processing and GIS

Digital image processing was accomplished using Leica's ERDAS Imagine version 9.1. Geographic Information System (GIS) vector and raster data analyses and map generation were accomplished using ESRI's ArcMap version 9.2 (build 1380).

RESULTS

The following project results are organized consistent with task deliverables identified in the BLM Statement of Work (SOW) found in the Intra-Governmental Order (IGO) Articles – FAI060021 Modification 3, an agreement between BLM and USGS EROS.

Task 1 – Paired Plot Greenness and Seeding Effectiveness

There is a need for low cost methods for monitoring vegetation trends in burned areas that are seeded versus those that are non-seeded. This effectiveness monitoring is necessary to determine if the seeding treatments significantly increased plant density and cover by comparing burned/non-seeded versus burned/seeded plot areas.

The most precise methods for monitoring seeding-effectiveness involve ground-based vegetation and soil seedbank sampling. However, these methods are expensive, which limits the spatial extent to which they can be applied. Remote sensing may provide a more cost-effective alternative, but only if the resolution and frequency of coverage is sufficient to detect vegetation trends that may occur among burned and unburned areas, and areas where different management treatments are applied. The Landsat series of earth imaging satellites represent a relatively low-cost option for applying remote sensing technologies to resource management issues. They have a good revisit cycle of 8 days, assuming an operational two-satellite constellation. Resolution for the Landsat ETM/TM sensors is 30 meters. A major plus for Landsat is the routine collection and archiving of overpasses for the USA. Other higher resolution (to sub-meter) satellite and airborne imagery sources are more expensive than (the now free) Landsat, but provide higher spatial resolutions that may be necessary to detect vegetation trends of interest. However, high resolution data is generally collected only on request, and therefore, imagery may not be available to combine with post-fire images to generate the dNBR burn ratio (dNBR requires pre- and post-fire images). Spectral band availability may also be an issue. Landsat sensors possess visible/infrared (VNIR), and short wave infrared (SWIR) bands that generally satisfy many vegetation mapping and monitoring

requirements. Many high resolution sensors possess VNIR bands, but lack the SWIR spectral band necessary for the calculation of dNBR.

The project study area, including parts of the Mojave ecosystem, provides a challenge for remote sensing and the detection of subtle vegetation characteristics regardless of sensor used. Low vegetation density and canopy cover results in remote sensing data dominated by non-green or non-vegetated components of the landscape such as dead vegetation, litter, soil and rock. As a first step in determining the optimal platforms and sensors for rangeland burn area and seeding monitoring, this task was designed specifically to assess the value of Landsat ETM/TM data at a resolution of 30 meters for burn area seeding and overall vegetation monitoring within the BLM Ely and Las Vegas Districts.

In 2005, BLM established 38 SNC plot pairs, with each plot 40-acres in size (Table 9-2). One of the two paired-plots was targeted for seeding and one plot was not seeded. Aerial seeding was completed in January and February of 2006. Field sampling was conducted by field crews in the summer of 2006, 2007 and 2008 for each of the 40-acre plots, and many other macroplot points across the SNC. For the remote sensing evaluation, 38 seeded/non-seeded plot pairs were compared. Aerial seedings evaluated in this effort were applied using three seed mixes tailored to mesic blackbrush, non-wilderness pinyon-juniper, and wilderness pinyon-juniper sites (Figure 9-4). In an attempt to monitor seeding effectiveness using a traditional field sampling approach, sampling design and subsequent analysis of the field data was conducted by USGS BRD staff, located at the Las Vegas and Yosemite offices. Differences among management treatments (seeded and non-seeded) in burned areas were evaluated. Evaluating the concurrence of remote sensing data with ground-based data is the focus of this task. If Landsat data appear to provide results consistent with ground-based data, then a future goal of determining if Landsat data is sufficient as a monitoring tool in the absence or reduced availability of ground-based data could be pursued.

Overall, the time series trend for both the wilderness pinyon-juniper (WPJ) and non-wilderness pinyon-juniper (NPJ) paired-plot sites appear similar in terms of the magnitude of the NDVI values and variation with time. The time series trend for the mesic blackbrush sites differed from the pinyon-juniper sites in terms of magnitude of the NDVI values and variation over time (Figure 9-5). The mesic blackbrush (MBB) sites had lower NDVI values and slightly offset peaks of greenness when compared to the pinyon-juniper sites. An exception occurred for the period of March 2008 (and potentially March/April of 2006; uncertain due to missing coverage), when the mesic blackbrush nearly equaled or exceeded the NDVI values for pinyon-juniper sites. These dates of high mesic blackbrush greenness appear to be consistent with the expected early green up period (March/April) of annual grasses in the mesic blackbrush sites.

Table 9-2. Paired-plot names (dp), seed mix type, acres/hectares and seeded or control (non-seeded) status.

demoplot	FID	ACRES	HECTARES	Seeded	Seed_Mix	dp
10	9	40.337	16.324	control	Mesic Blackbrush	mbb10c
48	47	40.818	16.519	seeded	Mesic Blackbrush	mbb10s
1	0	40.524	16.399	control	Mesic Blackbrush	mbb1c
39	38	40.184	16.262	seeded	Mesic Blackbrush	mbb1s
2	1	40.031	16.2	control	Mesic Blackbrush	mbb2c
45	44	40.392	16.346	seeded	Mesic Blackbrush	mbb2s
3	2	40.405	16.351	control	Mesic Blackbrush	mbb3c
40	39	40.356	16.332	seeded	Mesic Blackbrush	mbb3s
4	3	40.326	16.32	control	Mesic Blackbrush	mbb4c
46	45	40.255	16.291	seeded	Mesic Blackbrush	mbb4s
5	4	40.904	16.553	control	Mesic Blackbrush	mbb5c
41	40	40.407	16.352	seeded	Mesic Blackbrush	mbb5s
6	5	40.526	16.4	control	Mesic Blackbrush	mbb6c
42	41	40.352	16.33	seeded	Mesic Blackbrush	mbb6s
8	7	40.974	16.582	control	Mesic Blackbrush	mbb7c
43	42	40.758	16.494	seeded	Mesic Blackbrush	mbb7s
7	6	40.308	16.312	control	Mesic Blackbrush	mbb8c
47	46	40.302	16.31	seeded	Mesic Blackbrush	mbb8s
9	8	40.863	16.537	control	Mesic Blackbrush	mbb9c
44	43	40.061	16.212	seeded	Mesic Blackbrush	mbb9s
37	36	40.359	16.333	control	Non-Wild PJ	npj10c
75	74	40.708	16.474	seeded	Non-Wild PJ	npj10s
24	23	40.37	16.337	control	Non-Wild PJ	npj11c
62	61	40.337	16.324	seeded	Non-Wild PJ	npj11s
23	22	40.267	16.295	control	Non-Wild PJ	npj12c
65	64	40.686	16.465	seeded	Non-Wild PJ	npj12s
16	15	40.607	16.433	control	Non-Wild PJ	npj13c
58	57	40.555	16.412	seeded	Non-Wild PJ	npj13s
15	14	40.043	16.205	control	Non-Wild PJ	npj14c
60	59	40.266	16.295	seeded	Non-Wild PJ	npj14s
35	34	40.212	16.273	control	Non-Wild PJ	npj15c
73	72	40.092	16.225	seeded	Non-Wild PJ	npj15s
36	35	40.677	16.462	control	Non-Wild PJ	npj16c
74	73	40.337	16.324	seeded	Non-Wild PJ	npj16s
32	31	40.559	16.414	control	Non-Wild PJ	npj17c
70	69	40.215	16.275	seeded	Non-Wild PJ	npj17s
33	32	40.732	16.484	control	Non-Wild PJ	npj18c
71	70	40.723	16.48	seeded	Non-Wild PJ	npj18s
34	33	40.819	16.519	control	Non-Wild PJ	npj19c
72	71	40.558	16.413	seeded	Non-Wild PJ	npj19s
25	24	40.508	16.393	control	Non-Wild PJ	npj1c
67	66	40.139	16.244	seeded	Non-Wild PJ	npj1s
12	11	40.13	16.24	control	Non-Wild PJ	npj20c
63	62	40.46	16.374	seeded	Non-Wild PJ	npj20s
26	25	40.862	16.536	control	Non-Wild PJ	npj2c
49	48	40.411	16.354	seeded	Non-Wild PJ	npj2s
38	37	40.067	16.214	control	Non-Wild PJ	npj3c
76	75	40.333	16.322	seeded	Non-Wild PJ	npj3s
27	26	40.198	16.268	control	Non-Wild PJ	npj4c
68	67	40.878	16.543	seeded	Non-Wild PJ	npj4s
28	27	40.422	16.358	control	Non-Wild PJ	npj5c
52	51	40.3	16.309	seeded	Non-Wild PJ	npj5s
29	28	40.085	16.222	control	Non-Wild PJ	npj6c
51	50	40.374	16.339	seeded	Non-Wild PJ	npj6s
30	29	40.595	16.428	control	Non-Wild PJ	npj7c
50	49	40.664	16.456	seeded	Non-Wild PJ	npj7s
31	30	40.155	16.25	control	Non-Wild PJ	npj8c
69	68	40.223	16.278	seeded	Non-Wild PJ	npj8s
20	19	40.697	16.47	control	Non-Wild PJ	npj9c
53	52	40.909	16.556	seeded	Non-Wild PJ	npj9s
22	21	40.87	16.54	control	Wild PJ	wpj1c
66	65	40.091	16.224	seeded	Wild PJ	wpj1s
21	20	40.354	16.331	control	Wild PJ	wpj2c
61	60	40.178	16.26	seeded	Wild PJ	wpj2s
19	18	40.514	16.396	control	Wild PJ	wpj3c
55	54	40.838	16.527	seeded	Wild PJ	wpj3s
18	17	40.418	16.356	control	Wild PJ	wpj4c
54	53	40.047	16.207	seeded	Wild PJ	wpj4s
17	16	40.213	16.274	control	Wild PJ	wpj5c
59	58	40.097	16.227	seeded	Wild PJ	wpj5s
14	13	40.184	16.262	control	Wild PJ	wpj6c
57	56	40.743	16.488	seeded	Wild PJ	wpj6s
13	12	40.174	16.258	control	Wild PJ	wpj7c
56	55	40.665	16.457	seeded	Wild PJ	wpj7s
11	10	40.343	16.326	control	Wild PJ	wpj8c
64	63	40.061	16.212	seeded	Wild PJ	wpj8s

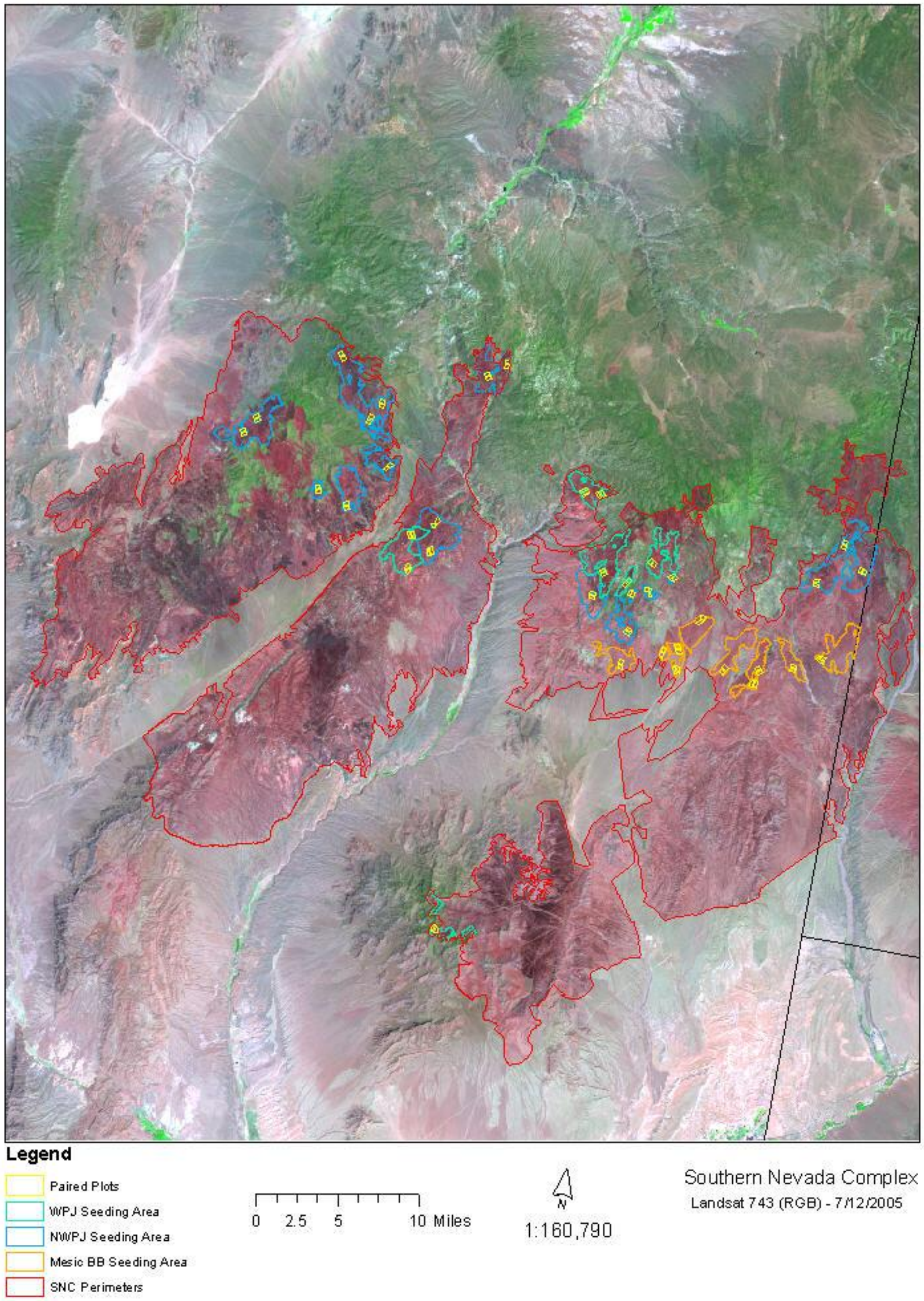

Figure 9-4. Location overview of SNC perimeters, three seeding mix areas (Wilderness PJ, Non-Wilderness PJ, and Mesic Blackbrush), and 40 acre paired-plot (PP) polygons.

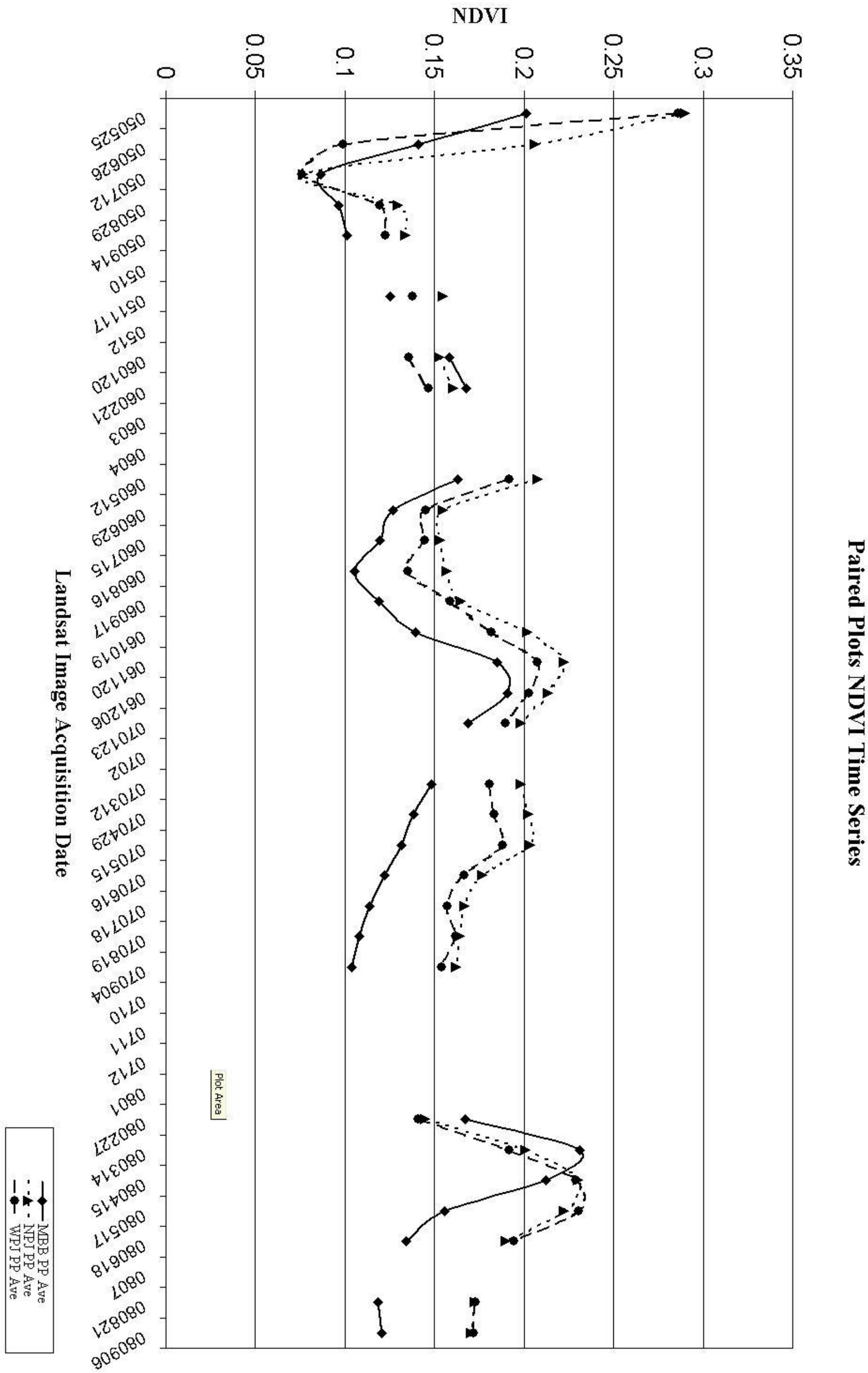

Figure 9-5. Average NDVI for all paired-plots (PP) within sites by seed mix type.

In 2006-2008, field data collected within the BLM SNC paired-plots indicated no significant establishment of seeded species and little effect by the aerial seeding (Chapter 5). This has primarily been attributed to low rainfall at critical times for the SNC seedings and possibly other factors related to competition from invasive annual plants and herbivory. This does not mean seedings could not prove successful in the future. However, given a project monitoring limitation of 3 years post-fire and the lack of seeding success in that period, the focus of this task required revision. Rather than focusing upon quantifying the level of seeding success (when there is likely little or no success), the evaluation simply looked at the spectral characteristics of individual paired-plots to see if they were consistent with field data, essentially showing no or very little variation between the seeded and non-seeded plots due to the aerial seeding. If plots could be identified that appeared to show seeding success (seeded plots greener than non-seeded), then either the remote sensing approach is identifying false seeding success or the field sampling data is not representative of the ground conditions. Given the rigorous nature of the ground observations, the first assumption would be the more likely. Other impacts or changes in the landscape due to factors unrelated to the aerial seeding effort (invasive species, annual and perennial plant dominance, etc.), will be addressed in following sections of this chapter.

NDVI values, representing levels of vegetation greenness, were extracted for the 40 acre plot polygons of the 38 paired-plots (seeded and non-seeded control) on a monthly basis for the period May 2005 through September 2008 (DVD - dpndvi_mean.xls). Based upon these data, approaches were used to determine if any paired plots exhibited characteristics suggesting seeding success: the seeded plot being greener than the non-seeded plot through time.

An assessment was conducted to identify if any of the paired-plots exhibited variation in NDVI through time, and if so, identify if any seeded plots show elevated NDVI greenness when compared to the corresponding non-seeded plots. Initially, differences between NDVI values for each paired-plot (seeded and non-seeded) were calculated for each time series interval excluding the pre-fire dates of May and June 2005 and pre-seeding dates of July 2005 through January 2006. Then these values (positive and negative) were summed across the remaining time series. From this information, plots showing greater summed greenness in the seeded plot than the un-seeded plot were identified. The data showed 19 plot pairs (50%) with an overall increased greenness level in the seeded plot versus the paired un-seeded plot when summed over the period of February 2006 through September 2008. To make this information more meaningful, a pre-fire comparison of paired-plot greenness was required.

Pre-fire baseline data only exists for May 2005. Using pre-fire information as a baseline, for the time period of 25 May 2005, 15 of 38 (39.5%) of later seeded plots showed greenness at greater levels than the corresponding non-seeded plots. One year later

(approximately 10 months post-fire and 3 months post-seeding), on 12 May 2006, 17 (44.7%) of seeded plots showed greater greenness levels than the corresponding non-seeded plots, an increase of two plots over the pre-fire period. Almost two years post-fire, on 15 May 2007, 19 (50%) of the seeded plots showed greater greenness levels than the corresponding non-seeded plots. Almost three years post-fire (2 years and 3 months post-seeding), on 17 May 2008, 19 (50%) of the seeded plots again showed greater greenness levels than the corresponding non-seeded plots. This represents a +10% increase in the number of paired-plots where seeded plots were greener than non-seeded. Of the plots showing greater May greenness in seeded than non-seeded plots, 11 of these plots showed this trend for the May date all 3 years. It is difficult and likely impractical to attempt to interpret this information or quantify it further. From a different perspective, one could also make the observation that half of the paired-plots showed greater NDVI levels in the non-seeded plots versus the corresponding seeded plot after over 2 years of monitoring. However, this result is consistent with the previous look at pair plots using a summation of greenness for the full post-seeding period. Both approaches suggest that in 2008 50% of the paired-plots were greener in the seeded plot and 50% were greener in the un-seeded plot, about the same or a slightly greater percentage than could be found pre-fire.

The magnitude of NDVI difference for each paired-plot may also be important. If a seeded plot is substantially greener than a corresponding non-seeded plot it may be more significant that a seeded plot marginally greener than a corresponding non-seeded plot. For May 2005 (pre-fire), a maximum NDVI greenness difference for plots showing greater greenness within the seeded plot was 0.081 for plot pair NWPJ20. All other plots showed a NDVI difference less than 0.05. Post-fire, in May 2006, all but two of the plots that showed greater NDVI levels in seeded versus non-seeded plots showed a difference of less than 0.05, with the two higher difference plots showing a difference of less than 0.067. For May 2007, the difference in NDVI values, for plots showing increased greenness in the seeded plot, was reduced with no plot showing NDVI difference levels greater than 0.03. This may suggest that plots showing increased greenness in seeded plots are only showing minor increases and potentially these differences are decreasing with time.

Overall, the paired-plot data shows NDVI variation through time with obvious highs and lows in NDVI response. Some plots exhibit a general increase in NDVI with time (Figure 9-6), while others show highest greenness in 2006 or 2007 with less greenness in 2008 (Figure 9-7). However, the two plots in each pair overall tend to exhibit very similar to near identical NDVI response (Figure 9-8). This suggests no difference in the response of seeded versus non-seeded (control) plots attributable to the influence of seeded species. No paired-plots showed an NDVI index difference greater than + or - 0.0775 for post-fire (12 July 2005 or later) dates.

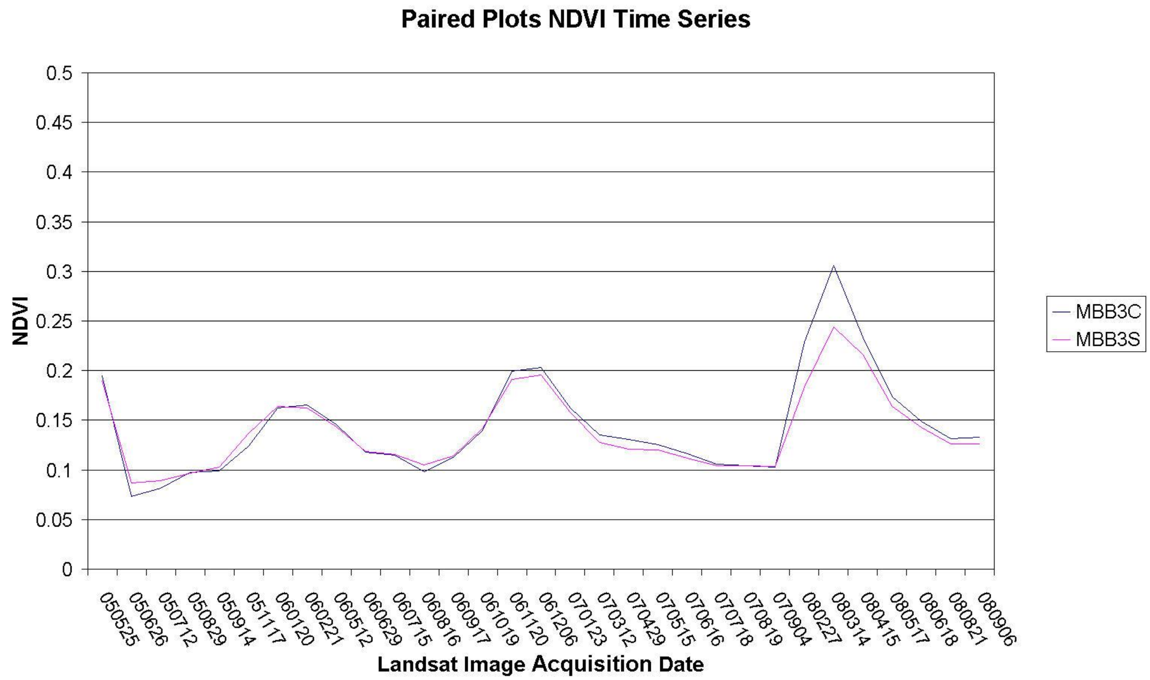

Figure 9-6. Comparison of mean NDVI values for seeded and non-seeded paired-plots.

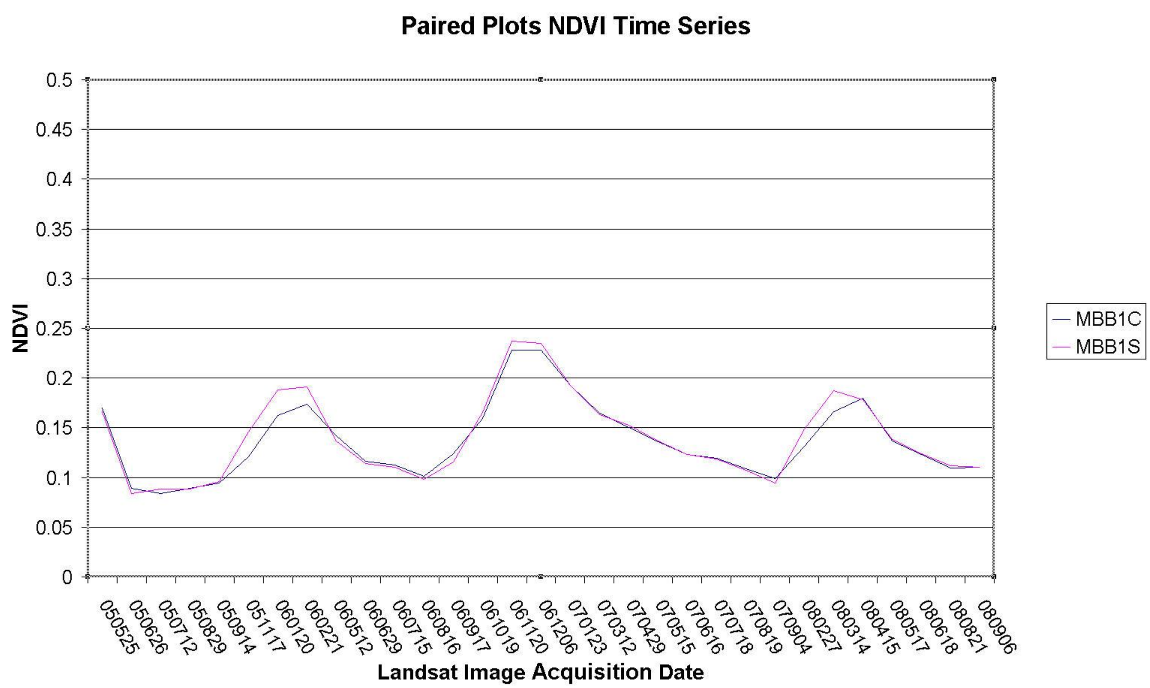

Figure 9-7. Comparison of mean NDVI values for seeded and non-seeded paired-plots.

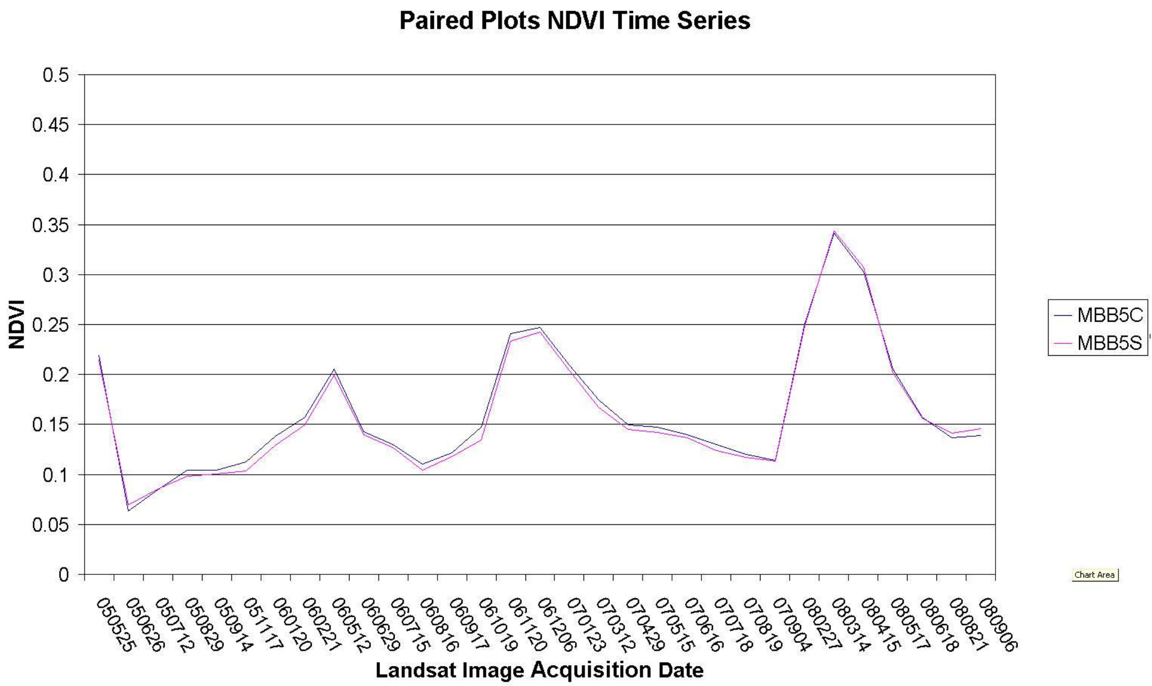

Figure 9-8. Comparison of mean NDVI values for seeded and non-seeded paired-plots.

Only one paired-plot (NPJ12) from the total of thirty eight paired-plots showed greater greenness in the seeded plot for all image dates. This observation was compared to the field macroplot estimates of the total live vascular plant percent cover or TVGC variable (that included perennials, annuals, and biennials) and was found to be consistent with the field derived data (Table 9-3) collected at several points within the paired-plots.

Table 9-3. The average TVGC (total live plant % cover) for all macroplots within seeded (S) and non-seeded/control (C) paired-plots by sample year.

DP	MPlot_ID	06TVGC	07TVGC	08TVGC
NPJ12C	MV15BB017	35.0	9.0	10.0
NPJ12C	MV15BB018	34.2	17.0	15.0
NPJ12C	MV15BB019	31.7	-1.0	-1.0
NPJ12C	MV15BB020	30.0	14.0	27.0
Total		130.8	40.0	52.0
Average		32.7	13.3	17.3

NPJ12S	MV15BB026	52.5	-1.0	-1.0
NPJ12S	MV15BB027	30.8	12.0	52.0
NPJ12S	MV15BB030	27.5	7.0	14.0
NPJ12S	MV15AA013	-1	32	40
Total		110.8	51.0	106.0
Average		36.9	17.0	35.3

In two cases, non-seeded (control) plots showed increased greenness for all image dates. These two observations were found to be consistent with macroplot point data (Tables

9-4 and 9-5) with only the exception of the average field point data for NPJ5S showing a slight (2%) greater TVGC value than the equivalent average field point for NPJ5C for 2007.

Table 9-4. The average TVGC (total live plant % cover) for all macroplots within seeded (S) and non-seeded/control (C) paired-plots by sample year.

DP	MPlot_ID	06TVGC	07TVGC	08TVGC
mbb4c	DU10BB011	12.5	12.0	31.0
mbb4c	DU10BB012	12.5	4.0	17.0
mbb4c	DU10BB013	17.5	73.0	16.0
mbb4c	DU10BB014	20.0	-1.0	-1.0
mbb4c	DU10BB015	7.5	-1.0	-1.0
Total		70.0	89.0	64.0
Average		14.0	29.7	21.3

mbb4s	DU10BB026	15.0	-1.0	-1.0
mbb4s	DU10BB027	18.3	12.0	17.0
mbb4s	DU10BB028	19.2	2.0	20.0
mbb4s	DU10BB029	13.3	4.0	15.0
mbb4s	DU10BB030	8.3	-1.0	-1.0
mbb4s	DU10AA037	10.0	-1.0	-1.0
mbb4s	DU10AA042	0.0	-1.0	-1.0
Total		84.2	18.0	52.0
Average		12.0	6.0	17.3

Table 9-5. The average TVGC (total live plant % cover) for all macroplots within seeded (S) and non-seeded/control (C) paired-plots by sample year.

DP	MPlot_ID	06TVGC	07TVGC	08TVGC
NPJ5C	DE17BB011	48.3	-1.0	-1.0
NPJ5C	DE17BB012	54.2	10.0	17.0
NPJ5C	DE17BB013	37.5	15.0	30.0
NPJ5C	DE17BB014	50	37	35
Total		190.0	62.0	82.0
Average		47.5	20.7	27.3

NPJ5S	DE17BB021	32.5	-1.0	-1.0
NPJ5S	DE17BB022	38.3	30.0	18.0
NPJ5S	DE17BB023	33.3	25.0	45.0
NPJ5S	DE17BB024	37.5	13.0	12.0
Total		141.7	68.0	75.0
Average		35.4	22.7	25.0

***t*-Test Of Paired Observations.** To further evaluate the effectiveness of the aerial seeding, we used a student t-test of paired observations for comparing the NDVI values extracted from the seeded and controlled plots. The t-test was performed separately based upon vegetation/seeding mix sites including MBB and a combination of NPJ/WPJ. The samples used in the test were selected from the Landsat image time series, with the t-test was run separately for these time periods:

- Pre-fire (only one date: 25 May 2005)
- All post-fire dates
- All greenness peak dates from 2005-2008 (i.e., 12 May 2006, 20 Nov 2006, 15 May 2007, and 15 Apr 2008)
- Individual greenness peak dates (12 May 2006, 20 Nov 2006, 15 May 2007, and 15 Apr 2008)

The null hypothesis of the test was that the mean of the difference in the NDVI vegetation index between seeded and controlled plots was zero. Table 9-1 to 9-6 presents results of the t-tests for each selected date (samples) by the seeding mix/vegetation type (MBB or NPJ/WPJ). All the tests show the p-values that are greater than 0.05. The tests failed to reject the null hypothesis at the alpha-level of 0.05. Thus, we concluded that there was no significant difference between the seeded and controlled plots in terms of the mean NDVI values.

Results from the greenness evaluation for paired-plots are consistent with field data analyses indicating insignificant seeding establishment across all seeded paired-plots (Chapter 5). The paired-plot greenness data appear to be consistent with the TVGC field plot parameter based upon a limited evaluation of TVGC and NDVI for selected paired-plots. This suggests the technique of monitoring greenness change within seeded and non-seeded (control) paired-plots using Landsat 30 meter data has potential for being a useful management tool. Based upon field data, paired-plots that exhibit very similar vegetation cover measurements on the ground exhibit the same similarity when observed with Landsat. With no successful seedings to add to the analysis, it can not be determined to what degree Landsat 30 meter data would be sensitive to vegetation greenness increases. Additional work is planned to compare 40 acre paired-plot average NDVI values to vegetation cover field variables averaged across all macroplots within paired-plot boundaries. A similar analysis, looking at all macroplots individually, is reported in TASK 6.

Table 9-6. *t*-Test of paired observations for vegetation indices extracted from non-seeded (controlled) and seeded plots. Mean of paired plot difference was based upon seeded minus control plot NDVI values.

Vegetation type	Image time serious or date	Number of observations	Mean of the Paired difference	Degree of freedom	*t*-value	*p*-value
MBB	All post-fire	300	-0.005	299	-1.59	0.1129
	Greenness peaks (060512, 061120, 070515, 080415)	40	-0.002	39	-0.39	0.7000
	060512	10	-0.005	9	-0.57	0.5826
	061120	10	-0.002	9	-0.15	0.8816
	070515	10	0.0000	9	-0.09	0.9338
	080415	10	-0.001	9	-0.33	0.7458
	Pre-fire (050525)	10	-0.038	9	-0.06	0.9498
NPJ/WPJ	All post-fire	834	0.0008	833	0.46	0.6481
	Greenness peaks (060512, 061120, 070515, 080415)	112	0.0014	111	0.26	0.7930
	060512	28	0.004	27	0.42	0.6777
	061120	28	0.0000	27	-0.06	0.9544
	070515	28	-0.002	27	-0.14	0.8873
	080415	28	0.004	27	0.73	0.4733
	Pre-fire (050525)	28	0.0000	27	-0.13	0.9010

Task 2 – Develop Prototype and Final Products

During the first two years of this three year effort, numerous prototype products were generated using remote sensing and field data from the 2005 through 2007 time period. These products helped refine the nature of the final products developed in 2009 using remote sensing and field data collected during the entire project (2005 through 2008). Prototype products included preliminary regression analyses, remote sensing data indices, other statistics, digital map layers, and hard copy map products. It is anticipated that all final project digital data including Excel spreadsheets, an Access database, numerous ERDAS Imagine images and derived raster layers, ArcMap project files and GIS data layers will be transferred to the BLM on DVD for archiving. Data used for map generation, PDF map files, and/or hard copy maps will also be transferred to BLM.

Task 3 – Acquire Satellite Imagery

Satellite imagery was purchased as originally planned with the exception of a few target acquisition dates where suitable Landsat scenes were not available due to clouds or snow cover in higher elevations. Data gaps occurred in periods considered both phenologically critical and non-critical (Table 9-1). A detailed description of the Landsat satellite image database developed for the SNC is provided in the DATA – LANDSAT TIME SERIES section of this report.

This SNC is located in an area (near the Mojave bioregion) that enjoys relatively low cloud levels, which allowed acquisition of a large number of cloud-free satellite scenes for use in this project. In other regions of the country that experience high frequencies of cloudy days, it would prove more difficult to obtain a similar time series or sequence of cloud free images. However, for future studies in other regions, as well as the SNC region, it may be possible to focus on obtaining only critical dates of imagery coinciding with critical plant phenological characteristics, thus reducing overall imagery needs. Continued availability of Landsat-like satellite data is critical to this kind of analysis in the future. NDVI appears an acceptable index for monitoring vegetation greenness in the SNC. Only VNIR spectral bands are needed to derive this ratio, and this enhances the possibility of obtaining comparable imagery from diverse satellites and sensors in the future. Additional satellite/sensor options would also provide increased observation opportunities. The Advanced Wide Field Sensor (AWiFS) may be a good example of a sensor with less spatial resolution (56 m compared to Landsat’s 30), but an increased swath width and frequency of overpass allowing for more temporal detail. MODIS imagery may also have value with high temporal resolution, but considerably lower spatial resolution (250 meter). A future approach combining Landsat with other lower spatial resolution data (50 meter to 1 km) sources that possess high temporal frequency (i.e., MODIS, AVHRR, and AWiFS) should be investigated.

Task 4 – Establishment of Seedings

The objectives of Tasks 1 and 4 are overlapping. The results for both tasks have been consolidated under Task 1 - PAIRED PLOT GREENNESS AND SEEDING EFFECTIVENESS.

Task 5 – Map Vegetation Recovery

The established SNC geospatial database provides an excellent tool to evaluate overall post-fire vegetation greenness and to estimate recovery in terms of the “return” to pre-fire or “background” greenness levels. Users must use caution and understand that this estimation technique does not specifically take into account how vegetation may have changed in composition or structure over time. Rather, it is just a comparison of the overall vegetation greenness to previous (pre-fire) levels. In many cases the landscape does change due to fire impacts and the potential establishment of non-native/invasive species. However, the

technique does provide a land manager with an overview of where the landscape is changing in terms of vegetation greenness. This change information may assist land managers in understanding or confirming/questioning other traditional data sources such as field data and expert opinion. The information can also be used to develop field sampling strategies that capture the range of variation within a study area.

A process to make assessments of overall SNC recovery by assessing the degree of the “return-to-background” (RTB) levels of NDVI greenness is described in detail under Task 11- “RETURN TO BACKGROUND” GREENNESS.

Mapping or quantifying vegetation recovery using NDVI can also be applied to management units such as grazing allotments. This can be done using data tables to display or describe results (Table 9-7), or by simply using interactive GIS or hardcopy maps of the management unit. An assessment of greenness recovery for all BLM Ely District allotments impacted by the SNC was completed and used to create a trend chart (Figure 9-9) and table (Table 9-6) showing recovery by allotment for the period 2005-2008. When examining these products, it is apparent that in general the majority of the allotments are responding in a similar manner through time. All allotments show highest NDVI greenness in the pre-fire year (2005). All allotments drop in greenness in 2006 after the fire. Perhaps surprisingly, all allotments again decline in average greenness in 2007. This may be attributable to more advantageous precipitation conditions in 2006 when compared to 2007. In 2008, all but five allotments increase in greenness to levels approximating the 2006 levels. Five allotments (Oak Springs, Delamar, Buckhorn, Grapevine, and Breedlove) decline in NDVI between 2007 and 2008 to their lowest May level between 2005 and 2008. Oak Springs and Buckhorn allotments were only slightly within the burn area. They lie to the northwest of the SNC and are dominated by the Delamar and Dry Lake valleys which are very arid. Delamar, Grapevine and Breedlove allotments are also located on the western margins of the SNC. The fact all five allotments are in close geographic proximity may suggest precipitation was an even greater limiting factor for these allotments when compared to the full burn area. Future effort will compare NOAA precipitation data with SNC NDVI data to understand these patterns in more detail. Additionally, comparing post-fire NDVI trends within allotments and using a baseline or pre-fire NDVI values from 2004 (or other previous years considered more representative of “normal” SNC conditions) may provide further insight.

Table 9-7. BLM grazing allotment mean NDVI, dNDVI (2005-2008), and percent NDVI values (2008/2005). The allotments are listed based upon pre-fire NDVI values. Note that no allotment has returned to pre-fire NDVI levels.

NDVI Mean Values in May 2005-2008

AllotNAME	2005 May Mean	2006 May Mean	2007 May Mean	2008 May Mean	dNDVI_05-08	% of 05 NDVI in 08
Barclay	0.312	0.292	0.265	0.285	0.027	91%
Sheep Flat	0.310	0.268	0.255	0.276	0.033	89%
Cottonwood	0.279	0.259	0.234	0.255	0.024	91%
Lower Riggs	0.274	0.227	0.180	0.215	0.060	78%
Pennsylvania	0.267	0.245	0.207	0.240	0.027	90%
Schlarman	0.244	0.200	0.171	0.209	0.035	86%
Boulder Spring	0.241	0.205	0.161	0.178	0.063	74%
Lime Mountain	0.239	0.192	0.150	0.187	0.052	78%
Rainbow	0.237	0.212	0.158	0.197	0.040	83%
Oak Springs	0.223	0.199	0.174	0.170	0.053	76%
Ash Flat	0.220	0.205	0.160	0.195	0.025	89%
Garden Springs	0.214	0.160	0.135	0.164	0.049	77%
Delamar	0.207	0.180	0.152	0.149	0.058	72%
Terry	0.202	0.157	0.115	0.144	0.057	72%
Snow Springs	0.202	0.152	0.120	0.147	0.055	73%
Mormon Peak	0.201	0.184	0.137	0.161	0.040	80%
White Rock	0.200	0.150	0.124	0.134	0.066	67%
Beacon	0.198	0.158	0.118	0.142	0.056	72%
Summit Spring	0.194	0.141	0.115	0.146	0.049	75%
Gourd Spring	0.183	0.157	0.116	0.138	0.045	75%
Henrie Complex	0.183	0.154	0.125	0.137	0.045	75%
Buckhorn	0.178	0.158	0.115	0.111	0.067	62%
Grapevine	0.174	0.151	0.120	0.118	0.056	68%
Breedlove	0.150	0.136	0.112	0.108	0.042	72%

This type of generalized vegetation (NDVI based) recovery product could be used as one of several possible factors in making grazing readiness determinations. At a minimum the NDVI return-to-background maps may be useful to land managers when they conduct on-site inspections to ensure the extremes of recovery status within a management unit are considered. The data could also be used to improve the selection and placement of monitoring/sampling sites to ensure the capture of the full range of variation within ecologically complex grazing allotments.

Task 6 – Map Post-Fire Annual Grass Dominance

NDVI values were extracted for all macroplots and compared to all remote sensing vegetation cover and vegetation density variables as previously described in section METHODS – FIELD DATA FOR REMOTE SENSING. Simple linear regressions and r squared values were used to evaluate each relevant date of NDVI data (i.e., 2007 images to 2007 field data) as a predictor of each of the macroplot variables. Additionally, in an attempt to improve results, the macroplot data was stratified by fire association and by elevation (above or below 4,300 feet).

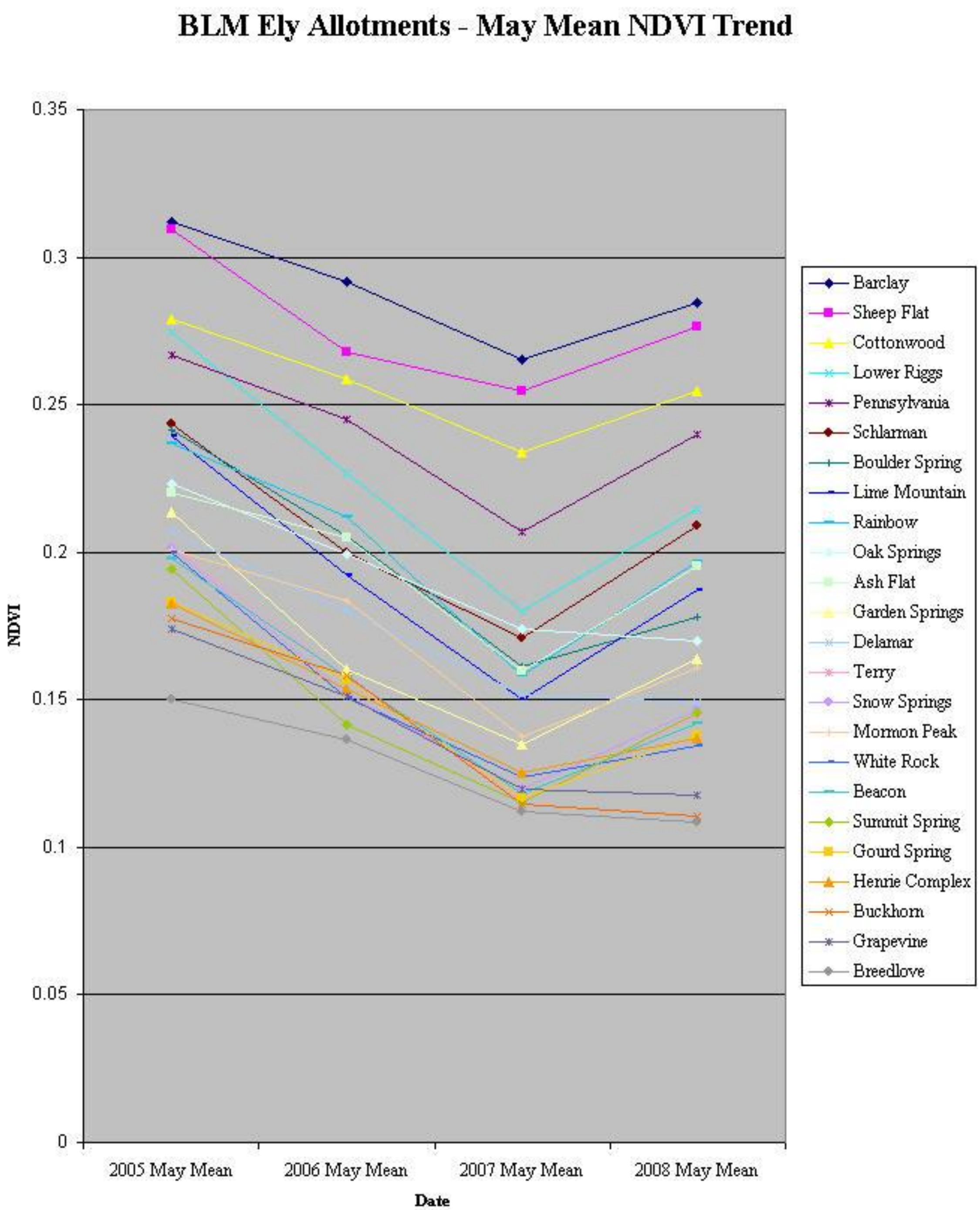

Figure 9-9. BLM grazing allotment mean NDVI values. The allotments are listed based upon pre-fire NDVI values. Note that no allotment has returned to pre-fire NDVI levels.

There was limited success in developing a technique for mapping post-fire patterns of annual grass dominance. The most encouraging regression results were isolated within the Delamar fire perimeter and limited to the variables 07_IAGC (total invasive annual grass cover) (Figure 9-10) and 07_BRTC (total cheatgrass cover) (Figure 9-11). The fact both

these variables had relatively similar results is not unexpected. 07_BRTC is a measure of total cheatgrass cover. Cheatgrass is an invasive annual grass, therefore it would inherently be included in the field measurement of variable 07_IAGC (total invasive annual grass cover). The regression equations associated with 07_IAGC (r^2 = 0.6056) and 07_BRTC (r^2 = 0.5111) were developed using field data from only the Delamar burn area. Reasonable annual grass % cover maps could likely be developed for Delamar but may not be suited for the full SNC area.

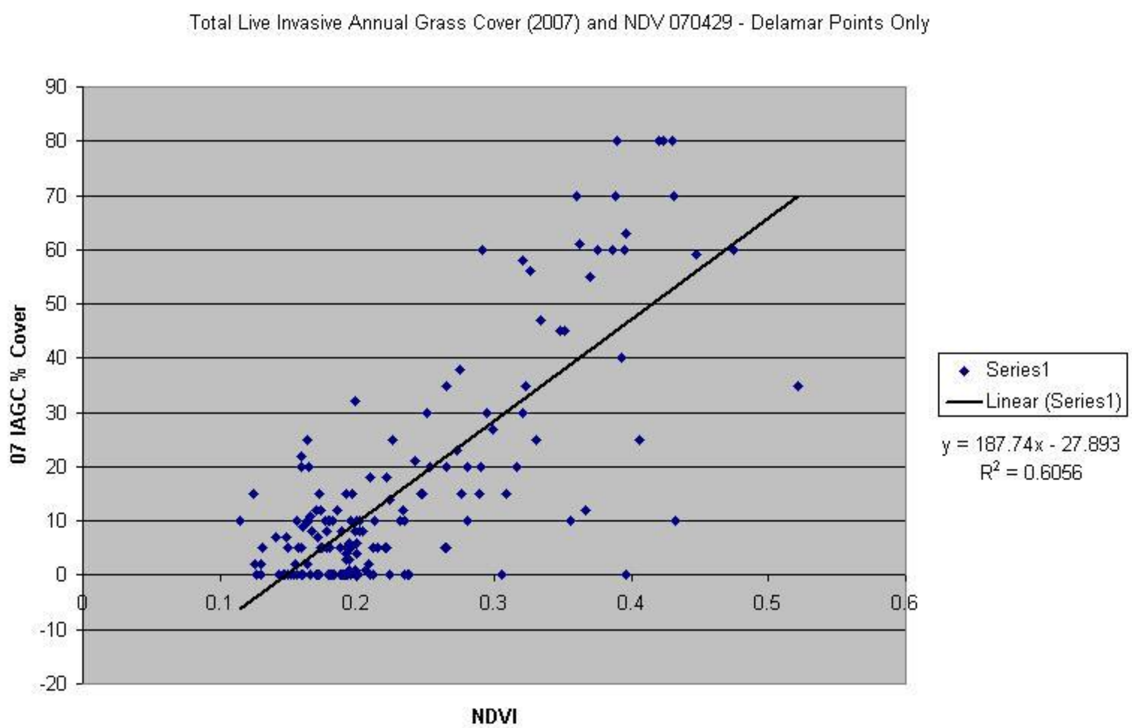

Figure 9-10. Regression results for 07_IAGC (total invasive annual grass cover) variable when limited to plots within the Delamar fire.

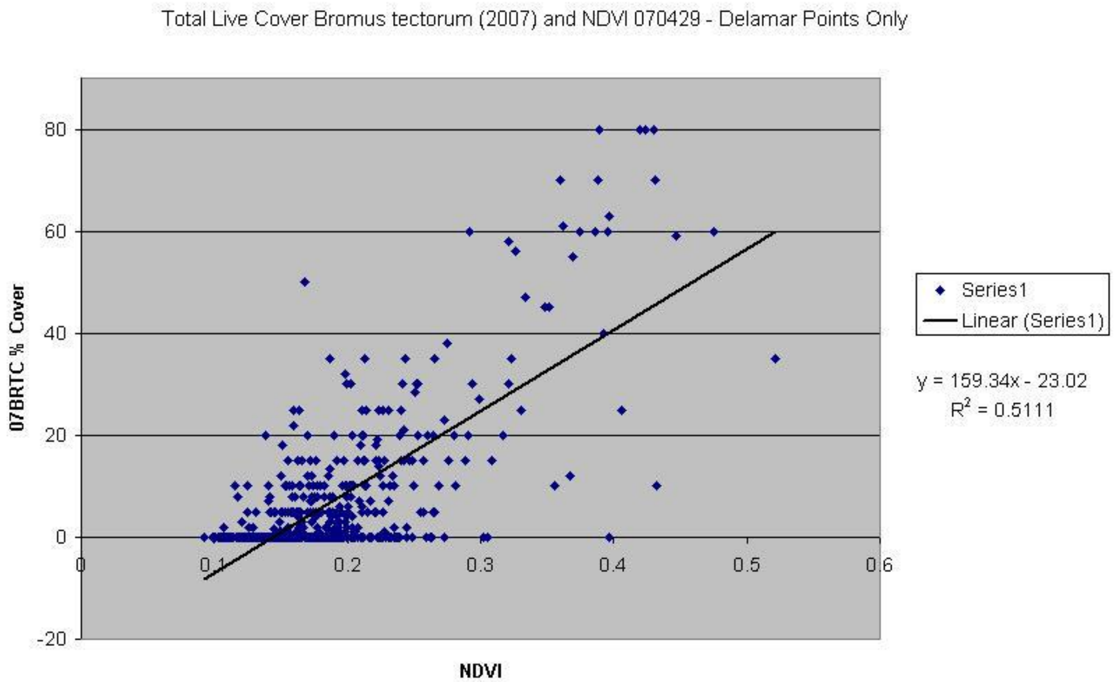

Figure 9-11. Regression results for 07_BRTC (cheatgrass total cover) variable when limited to plots within the Delamar fire.

In an attempt to develop an improved annual grass mapping methodology (better than the single date NDVI approach described above), a difference NDVI (dNDVI) approach was evaluated. This basic approach and other variations of this approach incorporating precipitation and temperature considerations have been attempted in several areas of the western United States with varying success (USGS. 2006; Bradley et al. 2005; Peterson 2003). A dNDVI based "cheatgrass ratio" has been suggested in this previous work and was also tested for the SNC. Cheatgrass ratio values for 2007 were generated by subtracting an NDVI image for non-green cheatgrass (070718) from an NDVI image near the peak of green for cheatgrass (070429). The values were extracted for all macroplot locations and again compared to the 07_IAGC variable. The resulting regression equation (Figure 9-12) and r^2 value (0.5097) were slightly less impressive than that achieved for the Delamar area with one NDVI date image, but this model has the advantage of being generated using field data from the full SNC area and therefore more suited for generating a full area map. The key to using the cheatgrass ratio appears to be tied to being able to select the best peak of green scene and non-green scenes for cheatgrass in any given time period. In a second attempt to map the SNC, using a slightly later NDVI image for peak of green (070515) and the same non-green scene used previously (070718), an r^2 of 0.4676 was obtained. The April 2007 NDVI image appears to be the image closest to peak of green for cheatgrass.

In the future, use of the cheatgrass ratio technique and other similar methods could be used to monitor the spread or occurrence of invasive annual grasses at relatively low cost.

Additional “high frequency coverage” sensors such as MODIS and AVHRR could also be applied to these efforts and perhaps used in combination with Landsat data.

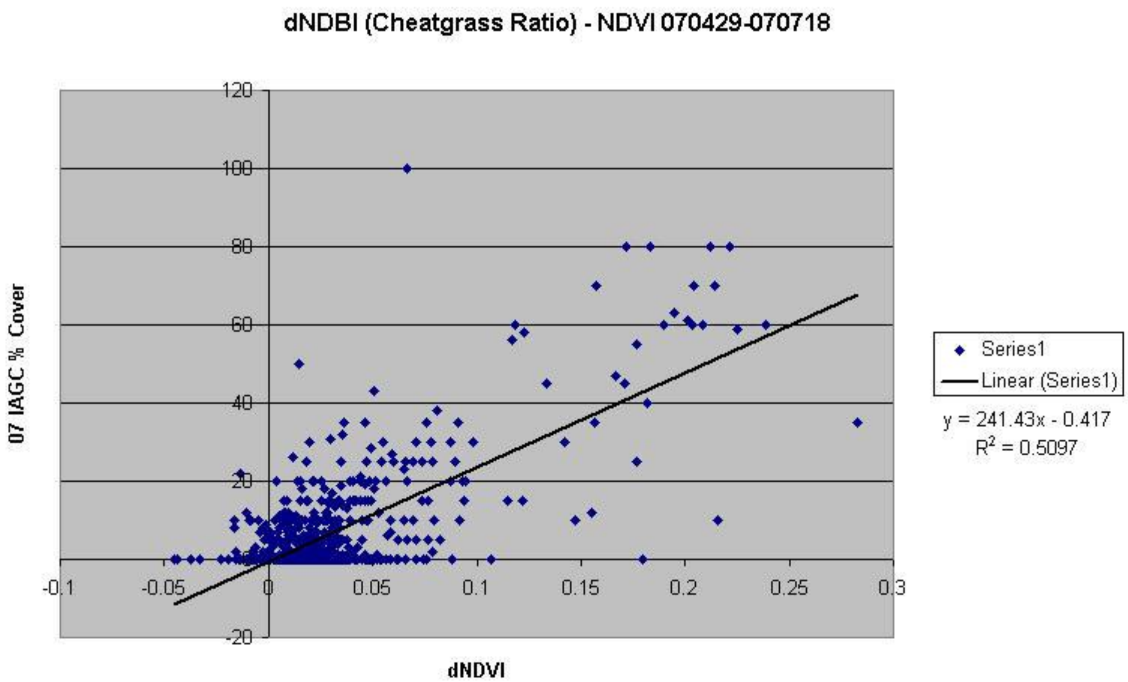

Figure 9-12. dNDVI Cheatgrass Ratio – Developed by subtracting non-green scene for cheatgrass(070718) from near peak green scene for cheatgrass (070429).

Task 7 – Map Post-Fire Perennial Dominance

The lack of macroplot field data representing areas of moderate to high perennial dominance made the mapping of the SNC area impractical. However, as with the work related to TASK 6, SNC macroplot data were compared to the NDVI time series. No reasonably strong r^2 values evolved from these efforts.

Task 8 – Develop Precipitation Index/GIS Layer

Precipitation data layers in ArcGrid format and providing nationwide coverage at 4 kilometer resolution were generated and delivered to the BLM Ely District. The data begin in January 2005 and continue through September 2008. January 1, 2005 was the initial date these data were created and distributed by NOAA NWS. This data continues without exception through September 2008, and if required by BLM, can be extended beyond September 2008 to the present. Details about the source of this data are described in the DATA – NOAA PRECIPITATION section of this chapter. The data were formatted to provide both daily (24 hour) totals as well as monthly summaries. This information was primarily used in conjunction with analyses of field data discussed in detail in Chapter 6 of this report.

Task 9 – Refine SNC BARC Using CBI and Other Data

An updated summary of soil burn severity by allotment (Figure 9-13) and BARC soil burn severity map (Figure 9-14) were generated for the SNC. The BAER team BARC map was originally based upon a 12 July 2005 post fire image and a pre-fire scene acquired prior to 2005 (06 July 2000 and others) in likely drier conditions. Due to unusually high rainfall and vegetation vigor in late 2004 and early 2005, it was decided that a pre-fire scene in the period immediately prior to the SNC fires (22 June – 10 July, 2005) may provide a more realistic assessment of pre- and post-fire conditions. A dNBR layer and BARC products were developed using a 12 July 2005 post-fire image and a 25 May 2005 pre-fire image. This revised BARC map was expected to be more highly correlated to soil burn severity and absolute biomass loss than it was to vegetation mortality. Eastern Nevada Landscape Coalition (ENLC) staff familiar with the SNC note that the revised BARC map likely contains confusion in low vegetation cover areas and specifically in the non-burn to low severity burn estimates. This is not unique to the SNC. Where a fine pattern of mosaic burning exists, often in very low vegetation cover sites, confusion discriminating non-burned and very low burn severity, with a 30 meter satellite pixel, is inevitable. Some of this confusion can be overcome by stratifying images by elevation, pre-fire vegetation types, etc. However, this process becomes complex and time consuming and may not be appropriate for BAER or other immediate response rehabilitation applications that tend to have compressed timelines.

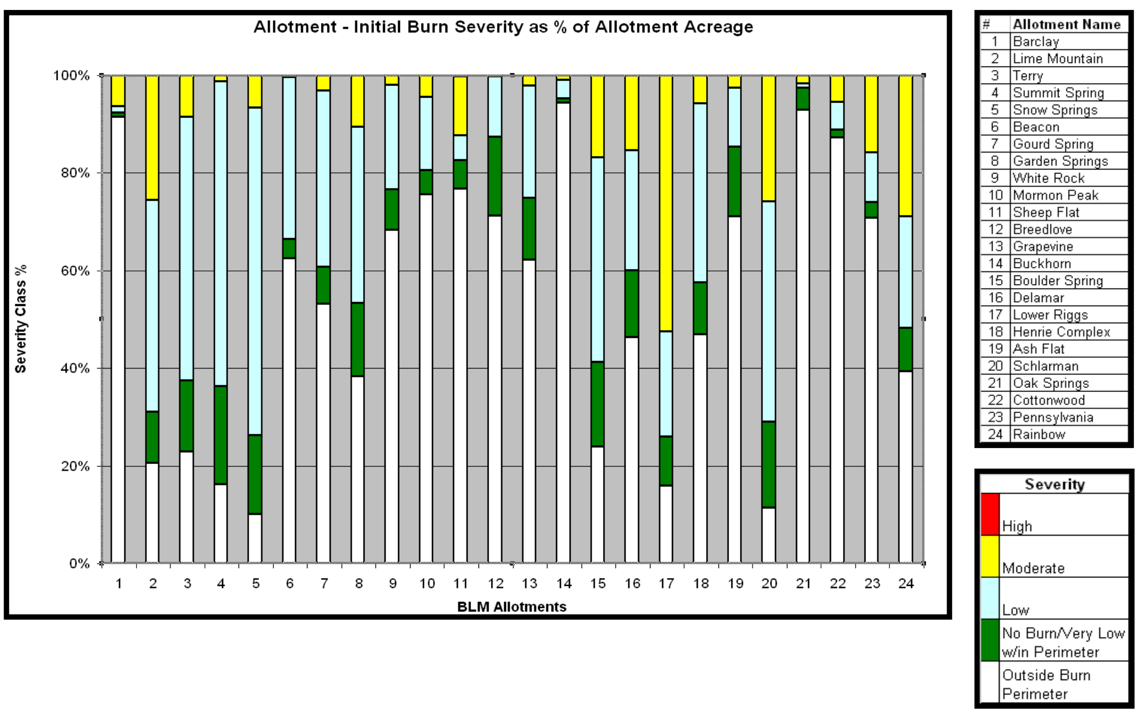

Figure 9-13. Immediate post-fire soil burn severity as a percentage of total allotment acreage.

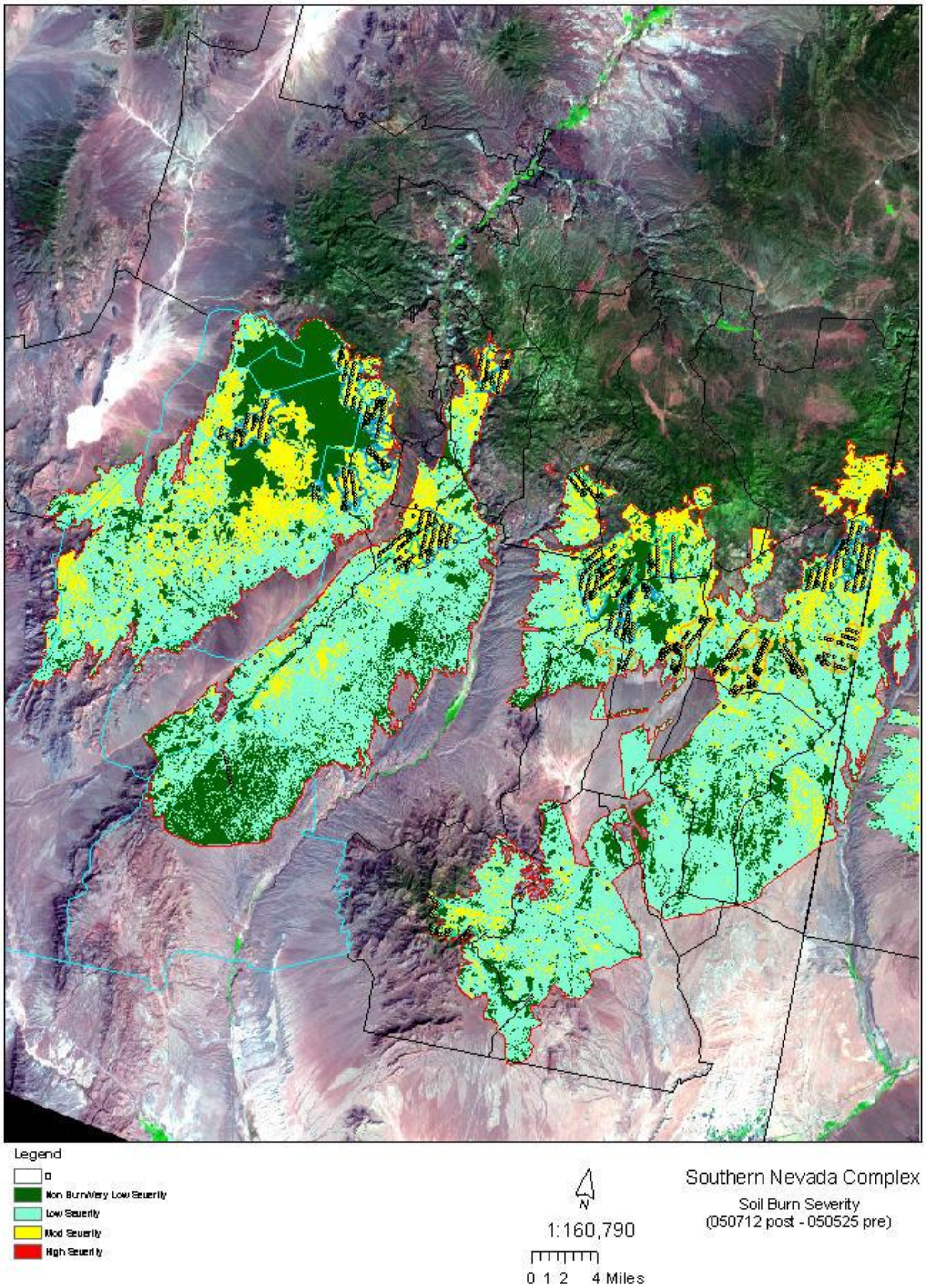

Figure 9-14. Immediate post-fire soil burn severity. Dark green areas are unburned. Turquoise areas are low severity, yellow areas are moderate severity and red areas are high severity.

A possible alternative to the BARC dNBR burn severity product is a similar product based upon the RdNBR index (Miller and Thode. 2007). RdNBR has been used to achieve good results in the Sierra Nevada Mountains and has shown some specific success in sparse vegetation (Miller and Thode. 2007). An RdNBR image was created for the full SNC using historical burn severity information derived from the USGS/USFS Monitoring Trends in

Burn Severity (MTBS) project (USGS/USFS. 2009). All MTBS mapped fires for the SNC were mosaicked into one SNC wide image layer and provided to the project team. Results related to vegetation analyses based upon field data and the RdNBR layer are provided in Chapters 4 and 6.

A comparison of RdNBR and dNBR (BARC) values to SNC Composite Burn Index (CBI) data is in progress. Issues with the full CBI point database have delayed this analysis but it is expected to be accomplished in the near future. Results from the comparison may not be conclusive, as RdNBR and dNBR have been compared in numerous study areas in many ecosystems across the western United States including Alaska. Results have been mixed for both approaches, with results described in several publications (Hudak et al. 2007; Lewis et al. 2007; Miller and Thode 2007; Miller et al. 2009).

A common issue with burn severity mapping is how to define "burn severity". Unfortunately, several definitions of burn severity exist depending upon user focus and needs. In general, users find dNBR to be more correlated to soil burn severity and absolute biomass loss, while RdNBR may have more strength as a measure of relative biomass loss or vegetation mortality. For the SNC and future fires in the BLM Ely District, each index may satisfy different information needs.

A comprehensive photographic database for all SNC macroplots was developed by BLM field contract staff. This database includes multiple photographs taken at the time the plots were visited by field crews in 2006, 2007 and 2008. The photographic database is maintained in Ely, NV by BLM along with the database containing the results of all macroplot field measurements. Macroplot dNBR, BARC256, NDVI and RdNBR values have been extracted for all macroplots. These remote sensing variables are maintained by USGS EROS with anticipated transfer to BLM. The composite burn index (CBI) was developed by Key and Benson (2005b). CBI is defined by the MTBS project as a numerical, synoptic rating calculated from a field-based estimate of fire effects on individual strata within a 30 meter circular plot. CBI estimates the overall impact to a site based on post-fire conditions averaged across the burnable portion of the site (USGS/USFS 2009). CBI ratings were calculated by SNC field crews at a subset of the macroplots visited in 2006. Design of a "key" to link these components (photographs, CBI, and burn ratio data) has not been completed. Further design discussions between USGS, BLM and the Eastern Nevada Landscape Coalition (ENLC) (BLM's partner in Ely, NV working under a cooperative agreement) should be conducted to finalize a design that meets BLM's needs. Effort could range from creation of a simple hardcopy document with references to all related data (photo, satellite indices, field parameters including CBI, etc.) to a more robust "GIS or Web based" application with more interactive or user query options.

Task 10 – Use Remote Sensing to Evaluate Plot Greenness
This task overlaps with Tasks 1, 4, 5 and 11. The evaluation of plot greenness with remote sensing data is covered in the responses to these tasks.

Task 11 – "Return to Background" Greenness
This task overlaps with requirements contained in TASK 5 – MAP VEGETATION RECOVERY. Results focused upon the recovery or "return-to-background" (RTB) NDVI levels summarized by BLM Allotments is provided under TASK 5. Results focused upon overall SNC recovery or "return-to-background" greenness follows.

An assessment of the degree that vegetation greenness has returned to pre-fire levels within the SNC was completed using a simple NDVI differencing methodology. Using the May 2005 NDVI image as the pre-fire image and May NDVI images from 2006, 2007 and 2008 as post-fire images, post-fire NDVI was subtracted from pre-fire NDVI (May 2005) to generate three GIS data layers and/or maps showing the dNDVI value. Values near zero indicate near pre-fire NDVI levels while larger positive values indicate a greater departure from pre-fire values or less vegetation. Negative values indicate the post-fire NDVI greenness exceeds pre-fire greenness levels (more vegetation). Digital maps can be adjusted to show the change as a continuous grayscale image or can be categorized based upon user needs (Figure 9-15). The current image database limits return-to-background estimates to the month of May as this is the only pre-fire image currently available to the project. In the future, additional pre-fire images are expected to be obtained, allowing for the generation of other pre- and post-fire comparisons.

Digital map layers contained on the transfer DVD provide return-to-background estimates for May 2006 (ndvi-dif_3934-35_050525less060512.img), May 2007 (ndvi-dif_3934-35_050525less070515.img) and May 2008 (ndvi-dif_3934-35_050525less080517.img) across the SNC. These maps can be adjusted to display user desired category thresholds or continuous grayscale. A small scale version of the May 2008, 2007 and 2006 return-to-background greenness products are also provided (Figures 9-15 to 9-17).

Task 12 – Description of Developed Techniques
Descriptions of significant techniques used to complete this project are found in the Methods section of this report. Tools such as ERDAS Imagine models, MS Access database queries, ArcMap project files, etc., will be transferred to BLM with appropriate metadata or user documentation.

To continue this type of remote sensing analysis for the BLM Ely District, specific software is required. This would include ERDAS Imagine or an equivalent image processing

software package, ESRI ArcMap (which the BLM already uses), and MS Access and Excel or compatible database and spreadsheet/statistical software.

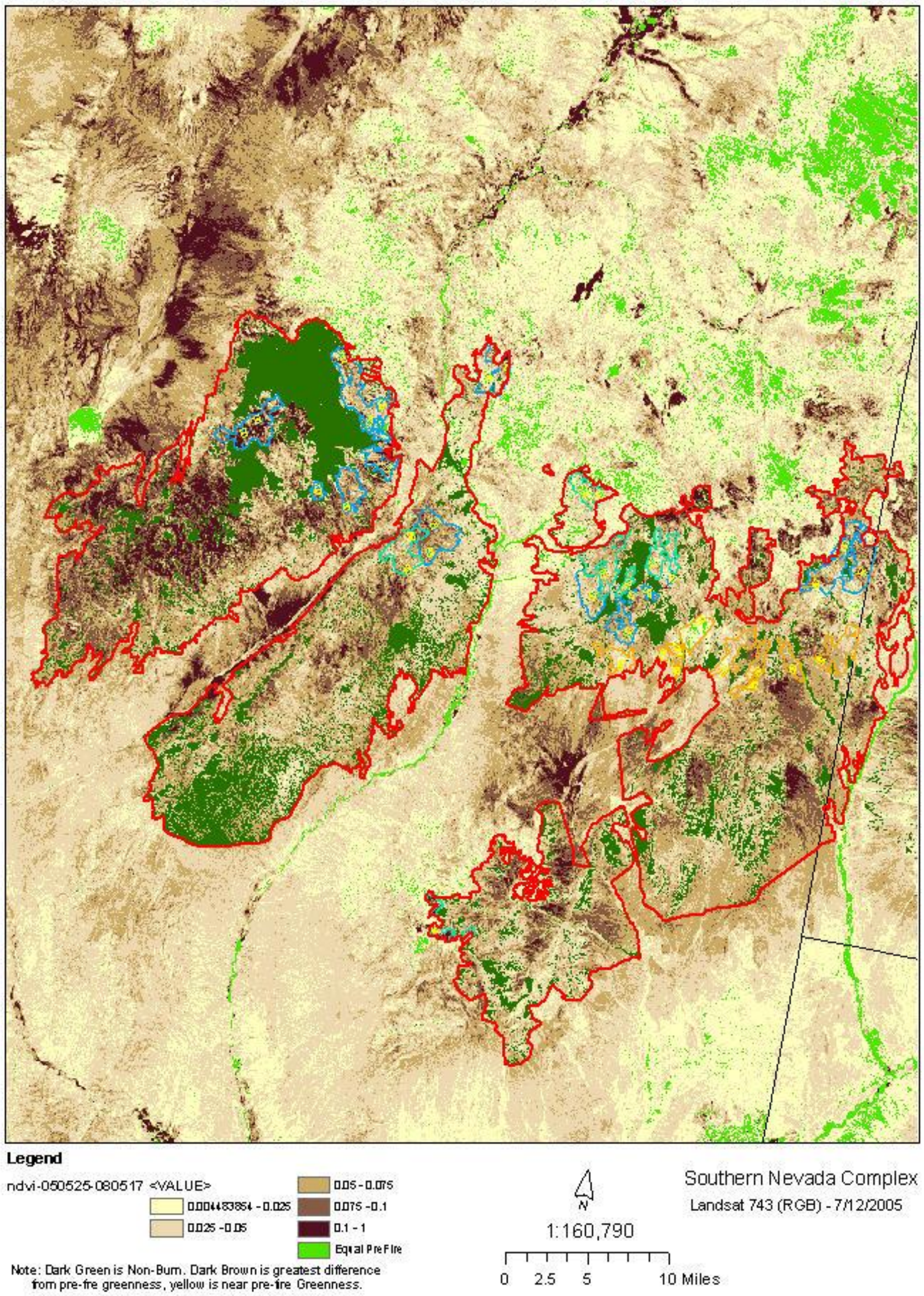

Figure 9-15. Return-to-background NDVI for 17 May 2008. This image shows the difference in NDVI for 17 May 2008 when compared to 25 May 2005. Note that a high proportion of the unburned area outside the fire perimeter is less green than in 2005. Burn perimeter and seeding polygon lines are provided for reference. Dark green areas are unburned.

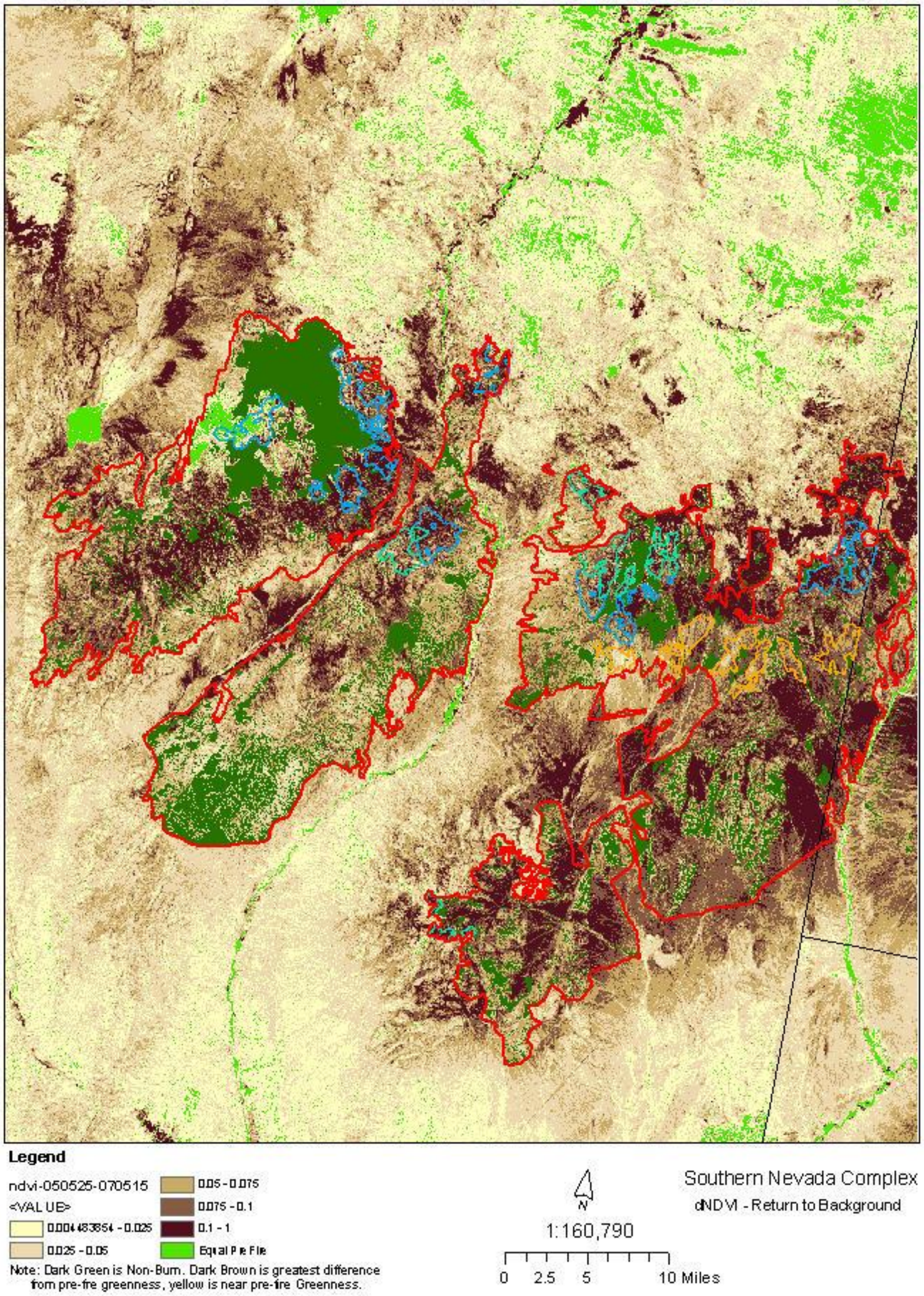

Figure 9-16. Return-to-background NDVI for 15 May 2007. This image shows the difference in NDVI for 15 May 2007 when compared to 25 May 2005. Note a moderate to high proportion of the unburned area is less green than in 2005. Burn perimeter and seeding polygon lines are provided for reference. Dark green areas are unburned.

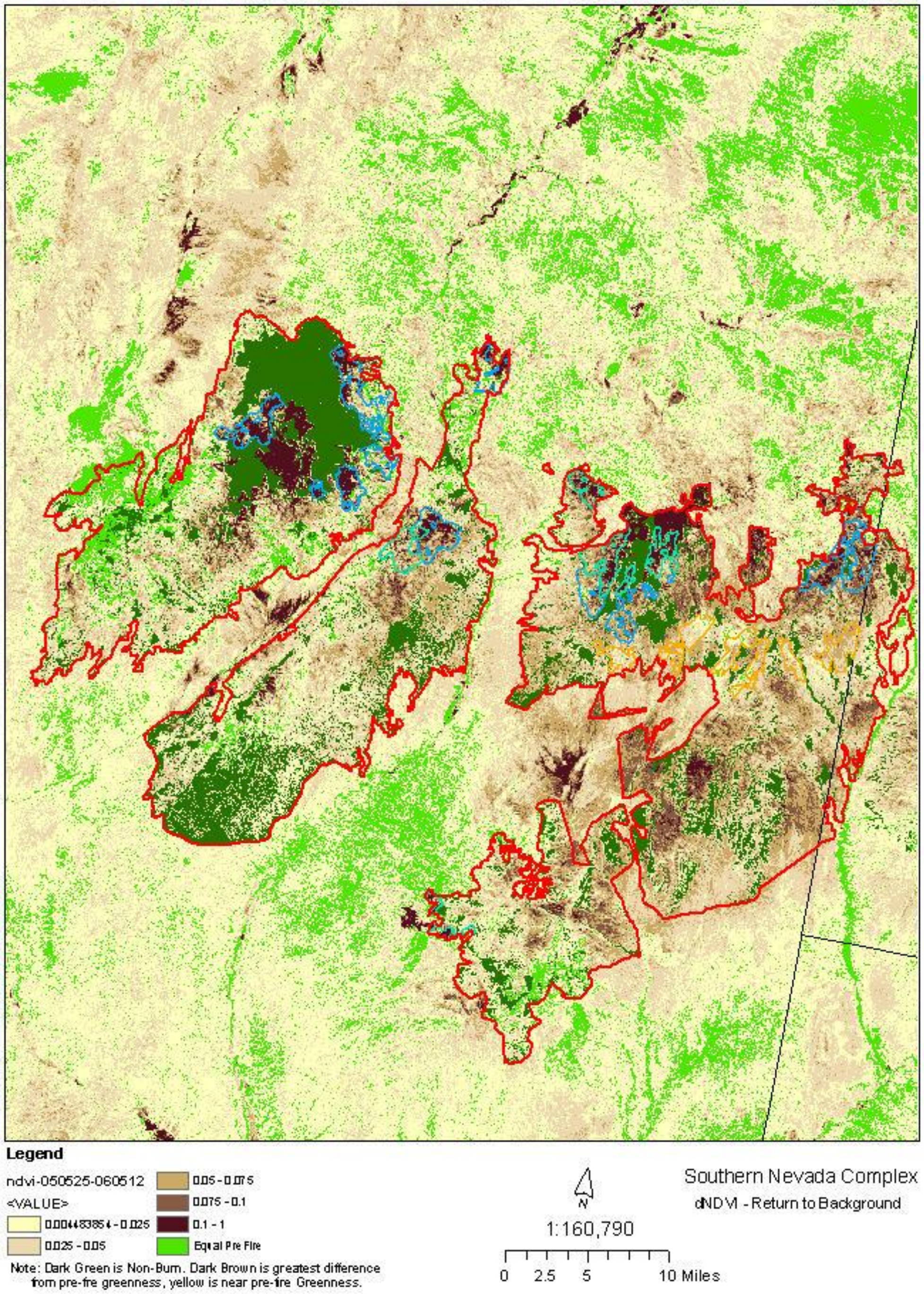

Figure 9-17. Return-to-background NDVI for 12 May 2006. This image shows the difference in NDVI for 12 May 2006 when compared to 25 May 2005. Note a lower proportion of unburned area is less green than in 2005, when compared to 2007/2008. Burn perimeter and seeding polygon lines are provided for reference. Dark green areas are unburned.

Other Results Not Specified in SOW
As this project evolved, many interesting spin off applications or analyses were identified. Some are research-oriented ideas while others are procedural or operational concepts that might lead to doing future analyses in a more efficient manner.

Seeding Location Selection. In support of BLM's seeding site selection process for the SNC, USGS EROS conducted BARC assessments immediate post-fire and later in the fall of 2005, for each of two Landsat path/rows (Path 39, Rows 34 & 35). Landsat 5 images were acquired, terrain corrected, reflectance corrected, and mosaicked to provide one digital image per acquisition date. Pre-fire and immediate post-fire images were used to map soil burn severity or the absolute loss of vegetation cover. Pre-fire and fall 2005 post-fire images were used to map recovery of vegetation cover. The vegetation loss and vegetation recovery map layers were provided to BLM so a higher priority for seeding could be placed upon areas where vegetation loss immediately post-fire was highest, and vegetation recovery by the following fall was lowest. Feedback from staff making the seeding location selections indicated these layers were useful, but not as much as anticipated. Other variables drove seeding site selection. However, USGS EROS had also generated fall 2005 NDVI greenness maps and tables showing greenness of potential seeding locations. These maps proved to be more useful to staff conducting the selection process. The fall 2005 NDVI greenness maps were easy to evaluate in the field when other logistical or field sites factors required shifting or locating new sites. The maps were also "current" in that they portrayed ground conditions at nearly the same time the crews were in the field. In the future, NDVI maps may be useful to staff making the seeding location determination. Developing these maps with data as near as possible to the actual date of on-the-ground seeding site evaluations would be best.

Selecting Macroplot Locations. The site selection staff were also provided with historical burn severity maps and perimeters identifying the location of recent (approximately last 10 years) burns in the immediate vicinity of the SNC. These maps, along with other BLM-generated historical fire perimeter maps, allowed staff to avoid establishing monitoring macroplots and seeding polygons in previously burned sites not easily detected in the field.

DISCUSSION

Landsat 30 meter remotely sensed data appear to be sensitive to seasonal fluctuations of annual grasses and perhaps perennial plants in the vicinity of the SNC. NDVI change over time, represented in tables and map products, including the return-to-background (RTB) product that assesses the degree a given burn area has regained or recovered to pre-fire vegetation greenness, appear to reflect known vegetation patterns and field data within the SNC burn perimeter. Periodic updating of maps and statistical summaries by grazing allotments is expected to continue through 2009. More evaluation is required, but these data

and related map products may prove valuable to land managers making “grazing or range readiness” determinations after large fires with limited quantitative information at a landscape scale. Future work to validate and field test this concept is anticipated. If successful, training for BLM Ely rangeland scientists and land managers concerning the use of the product would follow.

Given demand and applications that satisfy land manager needs, the existing Landsat time series database could be maintained into the future and enhanced historically at low cost. The recent “no cost” policy implemented by USGS provides free access to Landsat data including new acquisitions and archived historical images. This provides opportunities to continue SNC monitoring a minimal cost and to initiate monitoring for new seeding treatments in the future. No cost data will help to minimize time series “coverage gaps” during critical phenological periods by making multi-image compositing techniques, used to build cloud free coverages, more affordable. An expanded pre-fire SNC Landsat coverage is expected to provide new insights concerning fuel loads and vegetation characteristics prior to the SNC. Filling the Landsat coverage data gaps in the current database will improve on-going interpretations.

Work completed for the SNC has provided new insights concerning satellite image acquisition periods that capture critical phenological events in this ecosystem. It will be possible in the future to better anticipate critical image acquisition needs and to perhaps tailor image acquisition strategies more precisely, eliminating unnecessary acquisition and image processing costs. For example, knowledge gained as the direct result of the SNC project should help to repeat, and perhaps improve upon, promising results obtained using the cheatgrass ratio methodology for mapping invasive annual grasses in southern Nevada.

The lack of significant seeding establishment to date, for seeded plots within SNC paired-plots, has essentially eliminated the current opportunity to determine if Landsat data are useful for determining BLM seeding effectiveness in rangeland/arid ecosystems. The lack of unburned control plot data is also a limitation when attempting to extend SNC field information beyond burned area perimeters. Seeding effectiveness monitoring with Landsat data should be continued if possible beyond 2009, or at least until significant seeding success or substantial non-seeded vegetation regrowth is realized. It is recognized that practical limitations on continued field monitoring and data collection for the SNC may limit this option.

The SNC has a wealth of field data potentially suitable for analysis using advanced regression tree and decision tree approaches to land cover mapping. The lack of strong relationships between single date NDVI values and several vegetation cover and density variables could potentially be mitigated using methodologies that are more flexible and

compatible with a “database” approach to mapping. In general, Landsat-based land cover and canopy cover models derived using regression tree techniques have been found to be more robust than those derived by linear regression techniques. Tests are in progress to determine how applicable these techniques are to mapping burned area vegetation recovery, burn severity, vegetation mortality and other themes of interest to SNC land managers.

NDVI greenness mapping has potential for assisting in the future selection and/or evaluation of previously selected paired-plot (seeding and control) locations. NDVI can provide an indication of the similarity of vegetation productivity of these sites and in the case of plots in burned areas, can look historically in the pre-fire period to assess vegetation similarity. Additionally, post-fire burn severity (dNBR, RdNBR and dNDVI) can also be used to assist in locating paired-plots and other macroplots to ensure the capture of the range of variability across burned landscapes.

Burn severity assessments, including BARC mapping, have been shown to provide useful information for the SNC. The basic BARC suite of products provided assistance in identifying burned and unburned land within the SNC mapped perimeter. These “unburned islands” can be a factor in placing seeding and other post-fire treatments. Soil burn severity and vegetation mortality maps can be enhanced using Landsat burn severity mapping techniques. The role of RdNBR and dNBR in mapping burn severity within the SNC continues, with results expected in 2009.

NDVI greenness and burn severity map products assisted BLM staff in making seeding treatment location decisions for the SNC. This information, along with numerous other GIS data layers and logistical considerations, worked to improve overall decision making. The importance of including map products that were representative of field conditions at the anticipated time of proposed seeding site ground visits was reinforced by the SNC experience.

FUTURE WORK

As USGS EROS continues work on the SNC project and the associated SNPLMA effort during FY09, it is anticipated that additional progress will be made related to several tasks addressed in this report. In particular, the following objectives have been identified:

- Attempt to supplement the Landsat 5 image database to fill “gaps” that would significantly improve our understanding of the initial SNC time series. Specific dates likely to be considered are October and December 2005, March and April of 2006, and the October 2007 to January 2008 period.

- Extend the Landsat 5 image database in the pre-fire period. In particular, “peak of green” images for annual plants in 2004 may provide insight into the nature of SNC pre-fire vegetation characteristics leading up to the record setting 2005 fire season.
- Extend the Landsat 5 image database in the post-fire period from October 2008 to January 2009 to monitor expected winter green-up period.
- Add select Landsat 5 images to assist in characterizing pre-fire conditions in “normal” precipitation years.
- Explore methodologies to more fully compare time series NDVI values for paired-plots and macroplots, especially subtle seasonal differences over time that potentially relate to vegetation phenology differences due to annual and perennial vegetation in southern Nevada. Consider methodologies such as temporal smoothing of NDVI, Time Integrated NDVI, and other techniques and phenological interpretations usually associated with higher frequency coarse resolution satellite data.
- Continue assessment of immediate post-fire burn severity indices and SNC CBI data. Conduct a comparison of dNBR, RdNBR and NDVI values as they relate to CBI and other field variables indicative of soil burn severity and vegetation mortality.
- Continue periodic monitoring of seeding paired-plots, especially if local experts feel more optimal precipitation conditions exist that might lead to a significant seeding response. As well, continued monitoring would be justified if local experts feel conditions are optimal for increased dominance by annual plants.
- Continue return-to-background greenness mapping in May and late Fall 2009 for the overall SNC and summarize results by allotments for management consideration.
- Periodically browse 2009 Landsat imagery for the SNC. Investigate obvious landscape or vegetation pattern anomalies in coordination with local staff.
- Encourage revisits to a subset of SNC established macroplot sites, perhaps in some degree based upon a remote sensing determination of high change sites. RS plot types may be sufficient.

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